Empirical essays on Natural Resources and Environmental Economics

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Introduction

Natural resources are essential for human life since they allow the proper functioning of the surrounding ecosystems, providing adequate conditions for our existence (UN, 2020a,b). Furthermore, natural resources constitute the basis for human nutrition, the elaboration of diverse products and the provision of services (Caparrós-Martínez et al., 2022; Sagaya Jansi et al., 2021; FAO, 2020). Therefore, the cautious management and conservation of natural resources is of utmost importance in a moment in which some of these resources are highly exploited and negatively influenced by the climate change experienced (UN, 2015).

Fish is a natural resource used by humans for many different purposes. Its main applicability is direct human consumption. In 2018, over 87% of the global fish production was devoted to human consumption. Average human fish consumption has increased from 9 kilograms per capita and year in 1961 to 20.5 kilograms in 2018. In 2017, fish provided 17 percent of the global human animal proteins intake, representing more than half of the human animal proteins intake in some developing countries (FAO, 2020). Apart from human consumption, fish is used in other activities such as the elaboration of pharmaceutical products (de Andrade Belo and Charlie-Silva, 2022) and feeding farmed animals (EUMOFA, 2021; Alder et al., 2008).

Global fish production derives from two different sources: fisheries and aquaculture. The former produces fish by capturing it from its natural environment whereas the latter produces fish by farming it. The employment generated by these activities is not negligible. In 2018, over 59 million people were directly engaged in fish production at a global level, around 66% percent of them in fisheries and the remaining 34% percent in aquaculture (FAO, 2020). Nevertheless, the employment in fisheries is claimed to be underestimated due to the difficulties in tracking inland captures in some countries (Welcomme et al., 2010). The stagnation of catches partially motivated by the pressure of fish stocks exploitation (FAO, 2020; Perissi et al., 2017; Lotze and Worm, 2009) has put the attention into aquaculture as a complement to secure food availability. In the last decades, aquaculture has experienced a noticeable growth at a global level (increasing its production from less than 20 million tonnes in 1990 to over 80 million tonnes in 2018), especially in the Asian and American continents. However, around half of the global fish production still comes from fisheries. Indeed, fisheries are responsible of more than 80% of the fish production in 2018 in every continent except Asia (FAO, 2020).

The large employment generated by fisheries as well as its contribution to human consumption join to the environmental need of achieving a sustainable exploitation of these resources, especially nowadays when a significant portion of global fish stocks are already over-exploited. In particular, over 34% of global fish stocks were exploited beyond sustainable limits and almost 60% were exploited at the maximum sustainable yield in 2017 (FAO, 2020). Global initiatives have already been launched demanding countries to cooperate for the sustainability of fisheries (UN, 2015; CBD, 2010). Nevertheless, cooperation may be difficult when resources are unevenly distributed across parties (Hori, 2015; Klein et al., 2015; Halpern et al., 2013). The negative impact of unevenness in the distribution of resources on the cooperation for fisheries preservation has manifested from local (Fabinyi et al., 2015) to regional levels (Agnisola et al., 2019; Forse et al., 2019; O'Higgins and O'Hagan, 2019; Napier, 2016). At the same time, the implementation of certain global and regional policies targeting the improvement of the fisheries status has affected the distribution of resources across countries.

In the beginning of the 1980s, the UN restricted the access of countries to coastal fisheries to improve the conditions of their over-exploited fish stocks (UN, 1982). Thus, the management of waters within the 200 nm close to the coastline was assigned to the adjacent countries, which were the only ones with free access to these areas (Economic Exclusive Zones, EEZs). Fishing areas beyond the EEZs, high seas, remain freely accessible to every country. The implementation of the EEZs had significant consequences for many countries that were not allowed to access to the fisheries harvested freely previously and were forced to move and adapt their fleets to new fishing areas (Swartz et al., 2010). The increasing number of countries fishing in high seas rises the probability of over-exploiting these fisheries (Sumaila and Teh, 2015). Management policies beyond the regulation of Regional Fisheries Management Organizations (RMFO) are required to ensure the sustainable exploitation of high seas (Sumaila and Teh, 2015; Cullis-Suzuki and Pauly, 2010). In this respect, the analysis of the unequal exploitation of high seas by countries constitutes the basis for the effectiveness of the management policies (Sumaila and Teh, 2015). Chapter 1 analyzes the distribution of catches from high seas across countries in the last decades to frame the context in which countries are required to cooperate for the sustainability of public fisheries. This analysis has been already published as "Contributing to fisheries sustainability: Inequality analysis in the high seas catches of countries" in *Sustainability* (Gutiérrez and Inguanzo, 2019).

Within the European Union (EU), Member States (MSs) are committed to the Common Fisheries Policy (CFP), a framework regulating their fishing activity to achieve a sustainable exploitation of the European fisheries (including the EEZs of each MS, which are managed as a single area under this framework). Since its implementation in 1983, the CFP has been adapted to improve its efficacy. The last modification of the CFP in 2013 targets the maximization of the sustainable yield exploitation for all fish stocks, the minimization of discards, the adjustment of the fleet to the fishing opportunities, the development of the aquaculture as a solid alternative to fisheries and increasing the fisheries management based on scientific research. There are two types of measures used for these purposes: controlling the resources used in the fishing activity of each country such as the number of vessels or the engine power and controlling the catches obtained with fishing activity through quotas. Quotas for each fishery (Total Allowable Catches, TACs) are set annually according to the scientific advice on the stock status and are distributed across MSs following the Principle of Relative Stability, which prioritizes the fishing rights of MSs historically harvesting that area (EC, 2022; Breuer, 2022; EC, 2013). This distribution of quotas has been a concern for MSs (Matić-Skoko and Stagličić, 2020; Agnisola et al., 2019; Forse et al., 2019; Morton et al., 2016; Napier, 2016; McLean and Gray, 2009), who have even used different techniques to adapt their fishing rights to their needs (Hoefnagel et al., 2015; Morin, 2000). Chapter 2 aims to contribute to the design of a more equitable and sustainable distribution of catches across MSs to ensure the viability of the quota fisheries management of the CFP. To this end, Chapter 2 analyses the distribution of catches across MSs between and within the major EU fishing areas, searching for the differences in the use of production factors leading to the heterogeneity observed across MSs within fishing areas. This analysis has been already published as "The role of production factors on landings heterogeneity between EU countries" in *Marine Policy* (Inguanzo et al., 2021).

The implementation of policies targeting the sustainability of fisheries may also alter the distribution of resources within countries, having social and economic consequences. After the collapse of the Norwegian spring-spawning herring stock in the 1960s, Norway implemented fishing quotas for each vessel (Individual Vessel Quotas, IVQs) in the purse seine and deep sea trawler fleets to allow the recovery of the fish stock and prevent its future exhaustion (Hannesson, 2013). The later collapse of the Arctic cod in the 1980s motivated the implementation of the IVQs in the remaining Norwegian fisheries (Standal and Asche, 2018; Hannesson, 2013). IVQs are determined annually depending on the quotas established for each fishery (Total Allowable Catches, TACs) based on the scientific advice of the conditions of fish stocks. TACs are initially distributed across major fishing groups depending on variables such as the employment generated, the settlement and the efficiency in the use of resources. Quotas of the fishing groups are distributed across individual vessels considering aspects such as their size or tonnage (Norwegian Fisheries Directorate, 2015). Aiming to regulate the quota transactions across owners and facilitate the reduction of the fleet capacity IVQs were progressively allowed to be traded across similar vessels starting in 1996 with the purse seiners (Hannesson, 2013), continuing in 1997 with the deep-sea fleet and finishing between 2004 and 2007 with the coastal vessels (Standal and Asche, 2018). The implementation of catch limits as well as their commerce has direct consequences in the distribution of catches and revenues in the Norwegian fisheries. Indeed, Hannesson (2013) and Standal and Asche (2018) present evidences of an increasing concentration in the sector and an increase in the profits of the remaining vessel owners. Market concentration may not only have a negative impact on the consumer welfare, but may worsen the conditions of the labor market reducing the job opportunities and lowering wages (Hannah and Kay, 1977). Chapter 3 studies the evolution of the concentration in the value of catches across vessel owners in Norwegian fisheries in the most recent decades looking for its underlying factors. The motivation for this analysis as well as its core structure arose during the stay in the Fisheries Technology Department of SINTEF Ocean (Trondheim, Norway).

The unsustainable management of fisheries threatens global food security. In this respect, the catches discarded in the fishing activity are seen as a waste of resources that could help to ensure the future availability of food (FAO, 2020). Even though discarded catches may be reintroduced in the trophic chain by feeding other animals, these catches alter the balance of the ecosystem where they originate and the conditions of the environment in which are thrown and decompose (Clucas, 1997), aggravating the problems generated by the over-exploitation of fisheries regarding their sustainability and the food security. Therefore, several countries have already implemented unilaterally policies to avoid discarding practices in their fisheries (Condie et al., 2014). International regulations on discards are not currently implemented outside the EU, where the discarding regulation was proposed in the CFP modification from 2013 (EC, 2013). There are many reasons underlying the decision to discard catches by fishers, being the technical inefficiency and the low economic value of catches the most mentioned ones in the studies of particular fisheries (Maynou et al., 2018; Tsagarakis et al., 2014). Chapter 4 extends the previous analyses of particular fisheries to observe which factors motivate the discards produced by countries in order to design common policies targeting the minimization of global discards.

Fish demand gives the economic incentive to fishers for capturing and landing fish (van Putten et al., 2019). Besides the growth of global population, the increases in income occurred at a global level and the higher social awareness of the fish nutritional benefits have contributed to rising fish demand (FAO, 2020), intensifying the exploitation of fish stocks in an attempt to cover the increasing human fish consumption (Pincinato et al., 2022; Perissi et al., 2017; Frid and Paramor, 2012). Indeed, current food consumption patterns represent a threat to the environmental sustainability by magnifying the adverse variations observed in the climate conditions

(FAO et al., 2021; Mendenhall et al., 2020). In the case of fisheries, the climate variations experienced until the present date have already altered the conditions of the fish habitats, modifying the behavior of certain species and their development (Gaines et al., 2018; Tu et al., 2018; Lam et al., 2016). The food consumed by households entangles multiple environmental consequences from its production to its delivery and final consumption that may widen climate change and threaten food security (FAO et al., 2021). Among these environmental consequences, household food consumption is associated with the generation of greenhouse gas emissions (Aguilera et al., 2021), the use of water (Blas et al., 2019; Mekonnen and Hoekstra, 2012, 2011) and the waste of edible resources (Vázquez-Rowe et al., 2020; Gustavsson et al., 2011). The heterogeneity in the environmental impact of the different food categories has motivated numerous proposals of alternatives to current food consumption patterns according to specific environmental criteria (Esteve-Llorens et al., 2021; Blas et al., 2019; Castañé and Antón, 2017; Vanham et al., 2013). Price incentives have been acknowledged as one of the most desirable mechanisms to modify the behavior of consumers in order to decrease the environmental impact of food consumption and avoid large economic trade-offs (FAO et al., 2022). Therefore, Chapter 5 evaluates which are the optimal fiscal policies socially acceptable to minimize the environmental impact of food consumption in Spain regarding the greenhouse gas emissions, the water use, the food waste and a combination of these three.

During the elaboration of the present thesis COVID-19 arose. Acknowledged as a pandemic on the 11^{th} of March in 2020 (WHO, 2020), the spread of this virus has had enormous social and economic effects worldwide affecting fisheries and aquaculture as well (FAO, 2020). The special circumstances of confinement derived from declaration of the State of Alarm in Spain on the 14^{th} of March and the expertise in inequality metrics acquired with Chapters 1 and 2 motivated the application of this methodology to the understanding of the pandemic effects at a national level. Chapter 6 analyses how it spread in Spain during the first wave, focusing on the differences between and within the Autonomous Communities. This analysis was published as "Distributional impact of COVID-19: regional inequalities in cases and deaths in Spain during the first wave" in *Applied Economics* (Gutiérrez et al., 2021).

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Chapter 1

Contributing to fisheries sustainability: Inequality analysis in the high seas catches of countries [1]

The uneven exploitation of scarce natural resources threatens their sustainability by altering the commitment of agents. In fisheries, a great portion of catches is known to be concentrated in a few countries. Aiming to provide a more complete view on the distributional issues associated to the exploitation of common marine resources, this article focuses on the analysis of catches from high seas, which can be understood as the common marine resources under the current legislation. The analysis focuses on the evolution of several inequality indexes (the Gini index as well as others from the Atkinson and General Entropy families) from 1960 to 2014. Additionally, the Theil index is decomposed to observe whether this inequality is given by biological (*between* inequality) or technological (*within* inequality) reasons. All inequality indexes confirm that the exploitation of fishing resources in high seas is very unequal across countries. However, this inequality has decreased between 29% and 65% from 1960 to 2014. When considering the origin of catches, between 46% and 82% of the inequality observed is due to technological and fishermen capacity differences across the countries operating *within* fishing areas, while between 18% and 54% of the inequality can be attributed to biological differences *between* the fishing areas. Over time, the *within* component has decreased more than 35%, reflecting the greater reliance of more countries on high seas fisheries and their catching up on fishing technology. Being aware on the existence and the nature of catches inequality observed is necessary to develop successful policies for maintaining the sustainability of the fishery resources.

1.1 Introduction

Fish is a resource that is increasingly relevant to our lives. The United Nations Food and Agriculture Organization (FAO) estimates that fish accounted for nearly 17% of the animal proteins consumed by the global population in 2015. It is also widely utilized in non-food products such as fishmeal, fish oil, products for pharmaceutical uses and directly as raw material for animal feeds (FAO, 2018). Conserving it is, therefore, not only a matter of concern for the estimated 56.6 million people who depend directly on the sector, and international

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organizations are increasingly striving to preserve this natural resource. Indeed, the sustainable exploitation of oceans, seas and marine ecosystems has been set as one of the Aichi Targets for 2020 in the Convention on Biological Diversity (CBD, 2010) and as one of the Sustainable Development Goals established by the United Nations for the year 2030 (UN, 2015).

Aquaculture is becoming more important in the production of fish, but the amount of fish catches is still impressive. Indeed, in 2015, global catches exceeded 90 million tons, 170% higher than in 1960. The biggest increase happened from 1960 to 1990, and since then catches have remained almost stable. The decrease in production and subsequent collapse suffered already by several fish stocks are well documented (Perissi et al., 2017). The FAO estimates that the current exploitation of around 31% of fish stocks is unsustainable and warns that the production of fish will only increase if recovery programs for fish stocks are implemented successfully (FAO, 2016). Even if implemented, effects will not be immediate as the frequent time needed for the recovery of fisheries doubles and even triples the life span of species (FAO, 2018). Thus, preserving current ecosystems becomes crucial for the future availability of resources.

Following the concept of *Mare Liberum* (or The Freedom of the Seas) from Hugo Grotius (Grotius, 1609), the sea was viewed as a common resource free to all, so nobody could be denied its right to navigate or exploit it. Nevertheless, the need for preserving the marine resources supplanted this view by the concept *Mare Clausum*, which was reflected in the international law developments during the last century. After some countries unilaterally declared the ownership of their coastal waters after World War II, there was a need for regulating the property rights of the nearby coastline waters. This was formally encoded in the United Nations Convention in the Law of the Seas (UNCLOS, UN (1982)) which states that each coastal country has jurisdiction over the natural resources in the Exclusive Economic Zones (EEZs) defined as their adjacent 200 nautical miles waters.

After the restrictions imposed by the UNCLOS many countries lost part of the fishing grounds where they used to fish and, in some cases, they could only further fish in certain areas after buying fishing rights. The new status quo led to a clear expansion of fishing activities from the EEZs to the "high seas", i.e., to the international waters beyond the EEZs of coastal countries, which make up almost 60% of the oceans. As a result, the share of high seas fisheries on total catches increased from 0.73% in 1960 to almost 2.6% in 2014. High seas catches do not seem to play a large role in food security given the type of species caught and the main fishing entities benefited from these resources (Schiller et al., 2018). Nevertheless, the positive trend in the level of catches coming from high seas and in the amount of countries exploiting these areas (Sumaila and Teh, 2015) indicates that we must not disregard the relevance of these resources. Indeed, despite having slightly experienced the decrease observed in overall catches from 2000 to 2010, high seas catches have recovered and currently continue growing at an annual rate of almost 3%.

Despite the UNCLOS regulation, several studies suggest that the historical management of coastal fisheries has been unable to preserve fish stocks, resulting in strong negative impacts on coastal ecosystems (Pauly et al., 2002; Jackson et al., 2001). Consequently, most countries have developed recovery programs for the fisheries allocated in their EEZs. This is the case of the European Union (EU), which factored multi-annual recovery plans for certain fish stocks into the Common Fisheries Policy reform of 2002 in an attempt to promote sustainable fisheries management (Cardinale et al., 2013; Da Rocha and Gutiérrez, 2011; Hegland and Raakjaer, 2008). The United States also charged the US Fish and Wildlife Service with developing plans to recover the species listed in the US Endangered Species Act (USFWS, 1999).

Such policies are more difficult to implement on the high seas, which are governed by international entities formed by countries with fishing interests in an area, named Regional Fisheries Management Organizations (RFMO). Some of them manage all the fish stocks located in a specific area (e.g., Northwest Atlantic Fisheries Organization (NAFO)), while others focus on particular highly-migratory species across vast geographical areas (e.g., Indian Ocean Tuna Commission (IOTC)). Apart from the new legislation, several other factors have contributed to the transition from coastal to high seas fishing, e.g. the overcapacity of the fleets that led to the overexploitation of coastal waters (Palomares and Pauly, 2019), the technological development of the fishing vessels that made available the utilization of the deep-sea stocks (Morato et al., 2006; Roberts, 2002), and the government subsidization to the long-distance fleets that artificially increased their profitability (Sala et al., 2018). Overall, management policies implemented by RFMO to control the consequences of this expansion have been proven insufficient to prevent the depletion of high seas stocks. According to Cullis-Suzuki and Pauly (2010), two-thirds of the stocks fished on the high seas and under RFMO management are either depleted or overexploited.

Despite the interest shown for the trends in catches, research literature has paid little attention to distributional issues associated with fisheries exploitation of the common marine resources. Data on high seas catches reveal significant dissimilarities between the exploitation of different fishing areas and countries. Particularly, the data seem to indicate that high seas catches are concentrated in a small number of countries and fishing areas but this concentration decreased from 1960 to 2014. The new status quo defined by the UNCLOS in 1982 seems to be the main factor affecting this trend. Restrictions imposed by UNCLOS made that new countries started fishing in more productive fishing areas on the high seas; as time goes by, the more advanced technology required to access these fisheries becomes available for more countries.

Distributional concerns have focused mostly on income distribution. However, such concerns have recently been expanded to explain how the use of scarce natural resources and environmental capacity are distributed across countries. Azar et al. (1996) proposed a systematic framework of indicators for sustainability that focuses on just distribution of resources, not income. Although equitable distributions do not imply equal distributions, they suggest an indicator for intragenarational justice that compares the resources per capita used for a region (e.g., a country) to the amount per capita used for the world. Such an indicator implies that equality in the distribution of the resources is desirable for sustainability.

Traditionally, the management of international cooperation aspects has relied on the external imposition of actions and sanctions to countries. Nowadays, more flexible agreements are being developed to enhance the commitment of countries in the global sustainability. Nevertheless, accomplishing successful international cooperation in this frame requires the feeling of reciprocity, fairness and trust among participants. In this sense, heterogeneity in the distribution of the resources hardens the establishment of a common goal satisfying optimal conditions for all participants. This detriments the perception of fairness, reciprocity and trust, diminishing the willingness of countries to cooperate in international agreements for the conservation of the environment (Hori, 2015). Moreover, the link between the unequal use of the resource and the participation on cooperative agreement is bidirectional. Owusu et al. (2019) found that a non-cooperative behavior among the resource users drives to higher unequal harvests leading to a downward spiral of resource overexploitation and scarcity. Along the same lines, Drupp et al. (2018) concluded that valuation of nature should explicitly account for economic inequality and encompassing assessments of the distributional effects of environmental policies must consider the distribution of non-market environmental benefits. From the point of view of fisheries management, Fabinyi et al. (2015) found evidence, based on case-studies, that fishers are aware of and keen to act on resource sustainability. However, this predisposition is overridden by distributional concerns over who obtains benefits from the fishery. While more homogeneous distributions of resources may contribute to their sustainability by facilitating the accomplishment of agreements, the preservation of marine resources is affected by many other factors that should not be disregarded.

There is a large body of literature on the international distribution of natural resource use and environmental capacity. A non-exhaustive list includes articles about the distribution of CO_2, SO_2 and NO_x emissions (Azimi et al., 2018; Farrell, 2017; Duro, 2012; Padilla and Serrano, 2006), ecological footprint (White, 2007), energy intensity (Li and Jiang, 2017; Duro and Padilla, 2011; Alcántara and Duro, 2004) and material resources (Duro et al., 2018). In these studies, the traditional income inequality measures are applied for the analysis of distributional issues associated with resources or environmental goods. Considering fisheries as a scarce natural resource whose sustainability requires from the international cooperation, the present analysis extends this research approach to show the fishery resources distribution from the point of view of the exploitation. This may be of great relevance in designing policies aimed at preserving the marine ecosystem.

For this purpose, we focus on the distributional analysis of catches from high seas, which can be understood as the common marine resources *stricto sensus*, since the current legislation (UNCLOS, UN (1982)) on fisheries declared the EEZs as reserved areas to the respective country. The analysis is developed around two aspects. Firstly, the distribution of high seas catches across countries from 1960 to 2014 is studied using several inequality indexes. Secondly, the analysis seeks to learn whether the inequality observed in the distribution of high seas catches is due to the biological differences between the geographical areas where fleets operate or to the idiosyncrasies of countries (the type of fishing gears used, the characteristics of vessels, the amount of fishing labor, their preferences, the ownership of coastal fisheries, etc.). To address this second aspect, we make use of the properties of the Theil index, which enables inequality to be decomposed into different levels. To measure the use of fishery resources, Azar et al. (1996) proposed considering the population of countries as well as their catches. When considering the catches of countries jointly with their population, it is accounted the fact that larger societies may require more resources to cover their needs than smaller ones. As a result, the exploitation of fisheries from countries with heterogenous population can be compared avoiding the effects of the dissimilarity in their society sizes. For this reason, the distributional analyses implemented in this study weight the catches of countries by their population.

The rest of the paper is structured as follows. Section 1.2 presents the data used in this analysis. Section 1.3 details the methodology applied. Section 1.4 starts by looking at the main initial messages that can be drawn from the data. In particular, it overviews the trends in high seas catches, their relationship with the population and how they are distributed across the different fishing areas and countries. Once the context is analyzed, the evolution of inequality indexes and the Theil decomposition are presented. Finally, Section 1.5 presents our conclusions.

1.2 Description of data

To perform this analysis, we rely on the fishery catches provided by the *Sea Around Us* project (SeaAroundUs). This dataset provides reconstructed series of fishery catches that combine official reported data, mainly from the FAO, and reconstructed estimates of unreported data (including major discards) (Pauly and Zeller, 2015).

Apart from the corresponding fishing entities, this data allow distinguishing the areas where the catches occur. This is particularly convenient to address the analysis for high seas catches, which requires eliminating those captures coming from the EEZs regulated by the UNCLOS (UN, 1982) from the FAO fishing areas. Another advantage of this dataset is that it keeps the same countries along the period. The catch series have been reconstructed backwards for the countries that emerge from the break-up of old countries such as the Soviet Union, Yugoslavia, Czechoslovakia or Sudan. For the purpose of our analysis, we consider the catches from marine fisheries reported as landings from a total of 167 countries for which Sea Around Us offers data over the analyzed period, 1960 to 2014. Since we are looking for the unequal exploitation of a common resource across countries, all recorded catches from high seas are considered, independently of their future use (consumption or trade) and their monetary value.

The efficiency with which resources are being used to cover all the needs in a society is one of the principles that must guide any policy targeting the sustainability of such resources (Azar et al., 1996). Since it is not possible to attribute directly the use of fishery resources to each member of the society, the catches of countries are assumed to be equally distributed across their population. Technically, this is equivalent to weight the catches of countries by their population. Data on the population of countries is taken from the *World Development Indicators* of the World Bank (2019). In this regard, it is worth mentioning that the countries that are left out of the analysis, due to lack of data on catches, represent less than 10% of the global population.

1.3 Methodology

The present analysis aims to observe the evolution of the international high seas catches distribution in the last decades as well as to provide a broad explanation of the factors underlying it. For this purpose, several methodologies have been applied. Firstly, Lorenz curves for 1960 and 2014 are pictured in order to provide a graphic view on how inequality in the distribution of high seas catches changes from the beginning to the end of the period analyzed. Secondly, the evolution of the inequality during the period analyzed is quantified through several indexes. Following the suggestion of Duro (2012), a battery of inequality indexes is applied to achieve more robust results. Finally, we analyze how much of this quantified inequality can be explained by differences between fishing areas or by differences between countries within those areas. This distinction can be measured via the decomposition property of the Theil index. Bellanger et al. (2016) applied a similar approach to study the distributional effects of quota management on vessel production when the vessels are classified by subfleets or length classes.

1.3.1 Inequality Metrics

An easy way to show the dispersion of high seas catches around the world is to graph a Lorenz curve (Lorenz, 1905). This graph displays the information contained in a cross-tabulation of shares of catches and countries (or countries weighted by population). It relates the cumulative proportion of countries to the cumulative proportion of fishing catches, assuming that countries are arranged in increasing order of catches. A completely egalitarian distribution is represented by a diagonal line. The nearer the curve of the distribution is to this diagonal line, the more egalitarian the distribution is. The Lorenz curve is a powerful tool for inequality metrics because it enables the distributions of two populations to be compared. When the Lorenz curves of two distributions are displayed in the same graph and do not cross, it can be claimed unequivocally that the population with the curve closer to the diagonal is more egalitarian than the other. This claim can be extended to the case in which the Lorenz curves intersect under certain conditions.

Apart from this graphic analysis, distributional concerns can be measured objectively using inequality indexes. An inequality index can be understood as a distance function that aggregates the frequencies of a distribution in a particular manner. However, any inequality index fulfills four basic properties (Cowell, 2009; Atkinson, 1970; Theil, 1967): (i) anonymity (it does not matter which individual has each level of resources), (ii) population invariance (if the population is replicated, the inequality index does not vary), (iii) scale invariance (when a proportional change is applied to the whole distribution, the index reflects the same level of inequality) and (iv) and the Pigou-Dalton transfer principle (the index will show a decrease (increase) in inequality if an observation with more (fewer) resources gives part of them to an observation with fewer (more) resources).

Despite these basic properties, inequality indexes differ from each other in how they aggregate observations. Some indexes, such as the Gini index, are more sensitive to changes in the part of the distribution with more observations, which is usually around the mean (Allison, 1978; Atkinson, 1970). Others, such as the Atkinson or Theil indexes, may attach more weight in the aggregation to the values in the tails of the distribution. Therefore, what index is used depends on the issue to be addressed. Researchers interested in income inequality may lean towards the use of indexes that put more weight on the lower tail of the distribution; however, in environmental or natural resource applications, it may be more convenient to use neutral indexes (Duro, 2012). Table 1.1 summarizes the main characteristics of the inequality indexes most widely used in social science.

Note that the General Entropy indexes for the case of $\beta = 1$ (Theil index) and $\beta = 0$ (MLD index) requires applying logarithms to the level of catches. This is an important aspect for the analysis since many countries have zero catches in the high seas. Following the advice of the FAO (Bellù and Liberati, 2006), we consider that catches are equal to 1×10^{-25} tonnes for these cases to solve this deficiency.

1.3.2 The Decomposability of the Theil Index

When the individuals in a population can be classified in groups, it may be useful to decompose the inequality observed for the whole population into the inequality generated *within* the groups and the inequality due to differences *between* the groups. This is especially relevant in our study, where the catches by countries can be sorted according to the fishing areas where they were harvested. In this context, we are interested in learning what part of the inequality observed is due to differences within and between the fishing areas.

The Theil index is one of the measures that enables the inequality to be decomposed additively between and within groups (Shorrocks, 1984, 1980). When applied to our study, the decomposition of the Theil index can be formally expressed as:

$$T = T_{within} + T_{between},$$

being

$$T_{within} = \sum_{k=1}^{K} \frac{\sum_{i=1}^{m_k} n_{i,k} \cdot t_{i,k}}{\sum_{i=1}^{m} n_i \cdot t_i} \cdot T_k,$$

$$T_{between} = \sum_{k=1}^{K} \frac{\sum_{i=1}^{m_k} n_{i,k} \cdot t_{i,k}}{\sum_{i=1}^{m} n_i \cdot t_i} \cdot \left[\ln \left(\frac{n}{n_k} \cdot \frac{\sum_{i=1}^{m_k} n_{i,k} \cdot t_{i,k}}{\sum_{i=1}^{m} n_i \cdot t_i} \right) \right],$$

Table 1.1: Inequality indexes.

Formula *	**Main Characteristics**
	Gini Index (Gini, 1911)
$\frac{1}{2\,t\,n^2}\sum_{i=1}^{m}\sum_{j=1}^{m} n_i\, n_j\, \lvert t_i - t_j \rvert$	It is twice the area between the completely egalitarian distribution and the distribution in the Lorenz curve. Between 0 (egalitarian distribution) and 1 (maximum inequality). More sensitive to changes in the part of the distribution with more observations.
	Atkinson Indexes (Atkinson, 1970)
$1 - \left[\sum_{i=1}^{m} \frac{n_i}{n} \left[\frac{t_i}{t}\right]^{1-\epsilon}\right]^{\frac{1}{1-\epsilon}}$ for $0 < \epsilon \neq 1$	Parameter ϵ has to be selected from a normative point of view. It represents the social inequality aversion. The higher ϵ is, the more aversion to inequality society has.
$1 - \prod_{i=i}^{m} \left(\frac{t_i}{t}\right)^{n_i/n}$ for $\epsilon = 1$	Between 0 (egalitarian distribution) and 1 (maximum inequality). More sensitive to changes in the tails of the distribution.
	General Entropy Family Index (Shorrocks, 1984, 1980)
$\frac{1}{\beta\,(\beta-1)} \sum_{i=1}^{m} \frac{n_i}{n} \left[\left(\frac{t_i}{t}\right)^{\beta} - 1\right]$ for $\beta \neq 0, 1$	Parameter β has to be selected from a normative point of view. It represents the sensitiveness to the distance events at different parts of the distribution. The lower β is, the more sensitive the measure is to changes in the lower tail.
$\sum_{i=1}^{m} \frac{n_i}{n} \frac{t_i}{t} \ln\left(\frac{t_i}{t}\right)$ for $\beta = 1$	Between 0 (egalitarian distribution) and a value that depends on β and population (maximum inequality).
$\sum_{i=1}^{m} \frac{n_i}{n} \ln\left(\frac{t}{t_i}\right)$ for $\beta = 0$	The Theil index corresponds to $\beta = 1$ (Theil, 1967). The Mean Logarithmic Deviation (MLD) corresponds to $\beta = 0$.

* t_i and n_i represent catches and population of country i for $i = 1, 2..., m$, respectively; $n = \sum_{i=1}^{m} n_i$ is the global population; and $t = \sum_{i=1}^{m} \frac{n_i}{n} t_i$ denotes the global weighted average catches.

where (apart from notation in Table 1.1) K is the number of fishing areas, m_k is the number of countries harvesting in the fishing area k, $n_{i,k}$ is the population of country i attributed to fishing area k and T_k is the value of the Theil index calculated only on the population of group k.

Notice that the contribution of group k to total inequality, T, is given by $(\sum_{i=1}^{m_k} n_{i,k} \cdot t_{i,k} / \sum_{i=1}^{m} n_i \cdot t_i)T_k$. This term refers to the inequality *within* group k.

To apply the Theil decomposition to our study, it is necessary to attribute the population of each country to all the fishing areas where each country operates. To this end, we follow the equal distribution principle. By assuming an equal distribution of catches within the country, it is implied that the cumulative distribution of catches and population coincide. In other words, we are assuming that every percentage of the catches corresponds to the same percentage of population. Therefore, when the catches of a country are split by areas, by the same principle, the proportion of population attributed to each area corresponds to the share of total catches that they represent for the country. Formally, the population of country i attributed to the catches coming from fishing area k is defined as

$$n_{i,k} = n_i \cdot \frac{t_{i,k}}{t_i}.$$

Consequently, the population attributed to a fishing area can be estimated by adding up the population of all the countries that fish in it. Finally, adding up the populations of all fishing areas gives us the global population.

1.4 Results

1.4.1 Catches Evolution

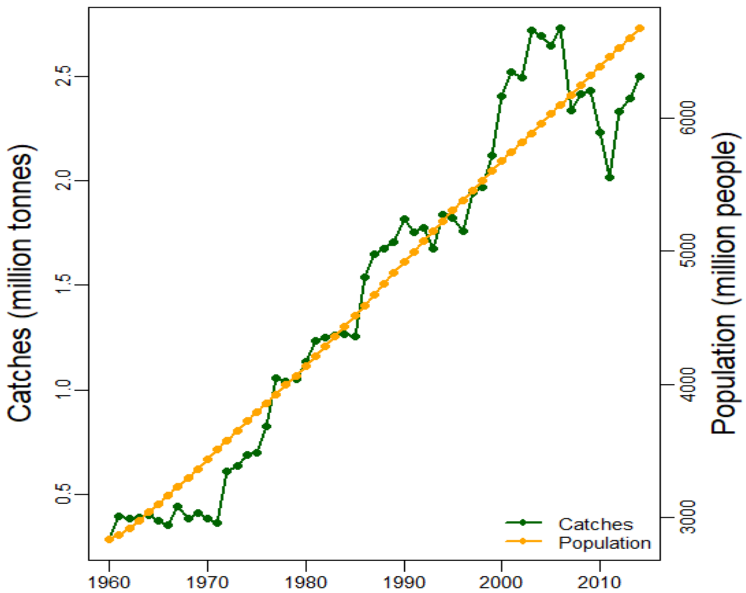


Figure 1.1: Evolution of high seas catches and population.

At a first glance, the data reflect that high seas catches have increased more than population from 1960 to 2014. During this period, catches have risen by around 784.19% with respect to their initial level, reaching the 2.5 million tons in 2014. As a consequence, the negligible share of high seas in the global catches of the 1960s (0.73%) has enlarged (representing 2.56% of global catches in 2014). There may be multiple factors underlying this increase such as the rise in population (global population grew from 3.03 billion in 1960 to 7.27 billion in 2014) (FAO, 2018), the technological advances in the fishing activity (Morato et al., 2006; Roberts, 2002), the overexploitation of the EEZs (Palomares and Pauly, 2019) and the governmental subsidization to the long-distance fleets (Sala et al., 2018). Contrary to global population, high seas catches have not increased steadily during the period analyzed. In Figure 1.1, which illustrates the paths followed by each variable from 1960 to 2014, this fact can be easily appreciated by the contrast between the steady line representing the population growth and the spiky one reflecting the path of catches.

Table 1.2: Compound annual growth rates* of high seas catches and population (in %).

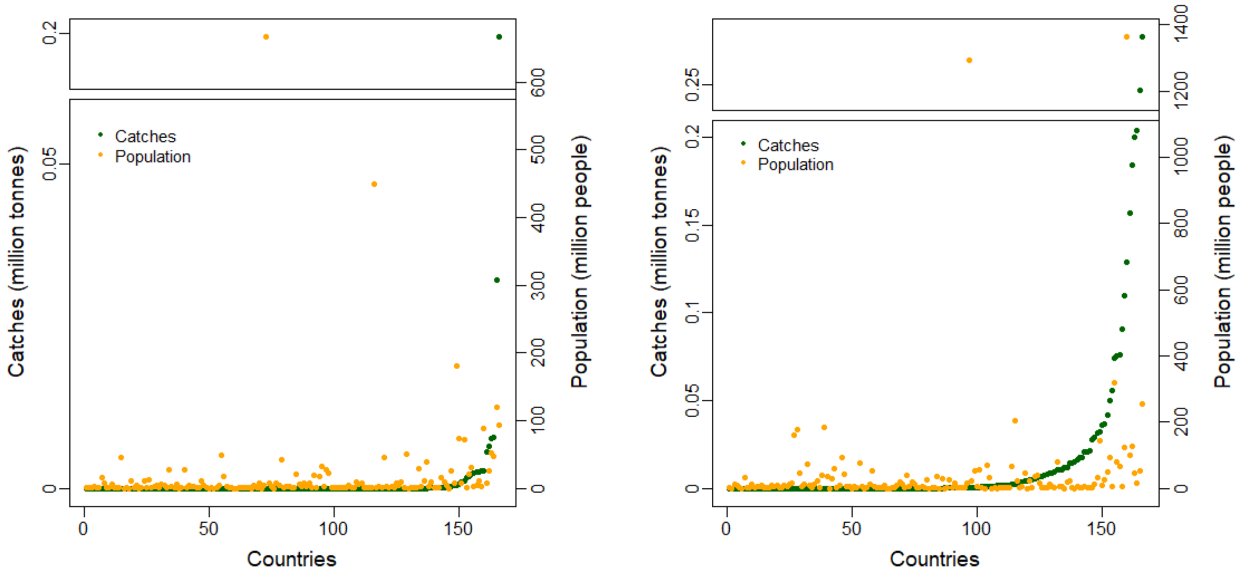
Period	Catches	Population
1960–1970	3.20	1.96
1970–1980	11.34	1.88
1980–1990	4.84	1.75
1990–2000	2.86	1.44
2000–2010	−0.76	1.20
2010–2014	2.87	1.11
1960–2014	4.12	1.61

* Calculated as $(t_e/t_b)^{1/n} - 1$ where t_e and t_b represent the value of the variable at the end and beginning of the period, respectively, and n is the number of the years.

Table 1.2 compares the evolution of population and catches focusing on their compound annual growth rates. Over the whole period analyzed, the compound annual growth rate of catches (4.12%) has more than doubled that in population (1.61%). This superiority of the catches growth rate seems to hold through almost all the decades (the only exception is the 2000s, where the catches even seemed to decrease). From 1960 to 1970, catches growth rate (3.2%) was already well above population growth rate (1.96%). During the next decade, catches growth increased spectacularly (11.34%), leaving the growth in population far below (1.88%). From 1980 to 2000, catches continued growing quickly with respect to population, but more slowly each decade when compared to their previous rates. In the next decade, catches growth rate became even negative (−0.76%), falling behind the population growth rate (1.2%). During the last four years of the period analyzed, catches seem to have recovered their previous positive growth rate (2.87%), overpassing again the corresponding population (1.11%).

1.4.2 Catches by Countries

Global catches are known to be heterogeneously distributed across countries (FAO, 2018). Figure 1.2 compares the amount of high seas catches and population of each country in 1960 and 2014. Aiming for a clearer representation, countries have been ordered by their level of catches. If fishery resources were evenly distributed across population, differences in the catches of countries would correspond to their demographic differences. Instead, it can be observed that dissimilarities in the catches of countries are not accompanied by demographic variations neither in the beginning nor at the end of the period analyzed. Nevertheless, Figure 1.2 clearly



(a) 1960

(b) 2014

Figure 1.2: Distribution of high seas catches and population by countries.

illustrates the increase in the number of countries participating from high seas fisheries during the period analyzed (Sumaila and Teh, 2015). In 1960, only a few countries fished in high seas. In fact, 80.12% of countries reported zero catches and only one country (Japan) reported catches above 100,000 tons. By 2014, the percentage of countries without fishing in high seas decreased to 48.19% while the number of countries reporting catches above 100,000 tons increased to eight (Indonesia, Korea, Ecuador and Spain are the top ones). From the inequality viewpoint, the distribution of catches from 2014 seems more equal as some countries that initially had zero catches from high seas ended up with a positive level of these catches.

Table 1.3: Shares of global fishing catches by quintiles.

Group	Share in Global...	1960	2014
1st quintile (less than 20%)	Catches	0.03	0.08
	Population	68.14	43.85
2nd quintile (20–40%)	Catches	0.34	0.76
	Population	1.05	4.22
3th quintile (40–60%)	Catches	1.91	4.25
	Population	12.97	7.33
4th quintile (60–80%)	Catches	4.91	11.68
	Population	2.50	1.56
5th quintile (80–100%)	Catches	92.81	83.23
	Population	15.34	43.03
Quintile ratio (S80/S20)	Catches	3093.67	1040.38
	Population	0.23	0.98

The evolution of inequality can be intuitively known by comparing the shares in catches and population of countries within different quantiles of the catches distribution in different years. Table 1.3 presents the shares in catches and population held by the countries within each quintile of the catches distribution in 1960 and 2014. As previously shown, this table reveals that countries in the lowest quintiles are not obtaining enough resources according to the population they represent. By contrast, the catches of countries within the largest quintiles over-represent their population. The quintile ratio provides an idea on how much larger are the shares in catches or population (depending on the case) of the countries in the last quintile with respect to those from the first quintile. Therefore, the closer is this ratio to 1, the more homogeneous is the corresponding distribution. In our case, the catches ratio is far above 1, reflecting that countries in the last quintile have levels of catches much larger than countries in the first quintile. On the contrary, the ratio for population is below 1, indicating that the share in population of the largest quintile is lower than that of the first one. Considering this, it can be known that catches per capita are not evenly distributed across countries. Nevertheless, the decrease in the catches quintile ratio and the simultaneous increase in the population ratio show that inequality in the catches per capita has fallen during the period analyzed. Even within quintiles, catches vary enormously from one country to another. In 1960, Japan alone accounted for 70% of high seas catches. The Russian Federation and France were the following countries with the largest shares of catches (11% and 3%, respectively). During the period analyzed, this concentration seems to have decreased extraordinarily. In 2014, Indonesia is the major fishing entity with 11% of the catches. Very close to this share are the ones from the Republic of Korea (with 10% of the global catches), Ecuador (with 8% of the global catches) and Spain (with 8% of the global catches).

1.4.3 Catches by Fishing Areas

If catches are disaggregated by fishing areas, their geographical concentration becomes clear. This information may be relevant for comparing the social pressure to which areas are exposed and propose a redistribution of fishing activity such that overexploitation of certain areas is avoided and global fisheries resources are more evenly distributed across population.

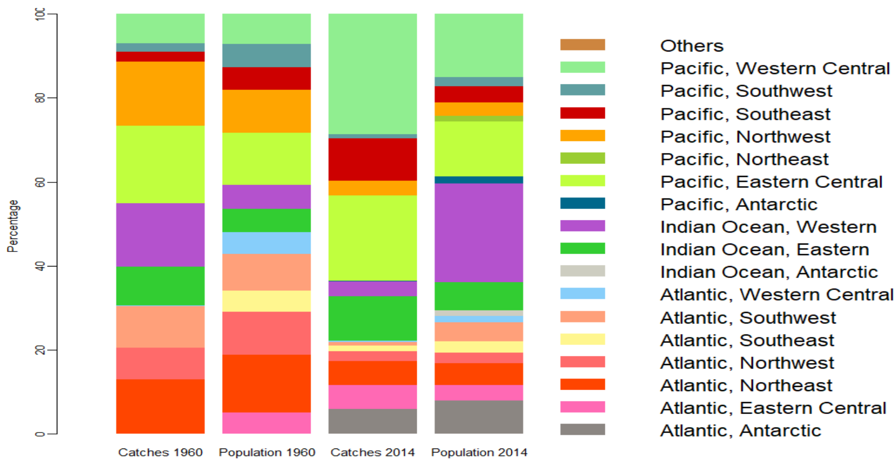


Figure 1.3: Contribution of fishing areas to high seas catches and population. “Others” represent fishing areas whose global catches and population were below 1% in 1960 and 2014.

There are only a few areas that make significant contributions to high seas catches, the rest account for a negligible proportion of the total distribution. Even though the share in total high seas catches of specific fishing areas may have changed over time, the shape of the distribution persisted from 1960 to 2014 (see Figure

1.3).

The Pacific Ocean represents the major source of high seas catches during the period analyzed, having increased its share in total high seas catches around 20% from 1960 to 2014. The Atlantic represents the second major contributor to high seas catches although it has experienced a decrease in its share of 8% from 1960 to 2014. These changes are in line with those observed by Sumaila and Teh (2015), who concluded that most countries have redirected their attention from species located in the North Atlantic to other species spread all around the world.

Within the Pacific, the most exploited areas correspond to the Eastern and Western Central Pacific. The former was the most relevant in 1960, but it is overpassed by the latter during the period analyzed. Both together accounted for 25.33% of the 1960 catches and 49.08% of the 2014 ones. Japan was the main fishing entity operating in both areas in 1960 with 85% of its catches. The prevalent position of Japan vanished over time. In 2014, countries such as Mexico and Ecuador become the major fishing entities in Eastern Central Pacific and Indonesia, the Republic of Korea and Philippines in Western Central Pacific.

The asymmetry in the number of countries and the population within fishing areas explains some of the heterogeneity in their catches. Nonetheless, considering the proportion of the global population represented in each area fosters the notion of inequality in their catches. As can be observed in Figure 1.3, there are some areas whose contribution to global catches is well below their share in global population (such as the Atlantic or the Indian Ocean in 2014), whereas other areas have shares in global catches larger than their percentage in the global population (such as the Pacific Ocean or the Indian Ocean in 1960).

1.4.4 Global Inequality in Catches

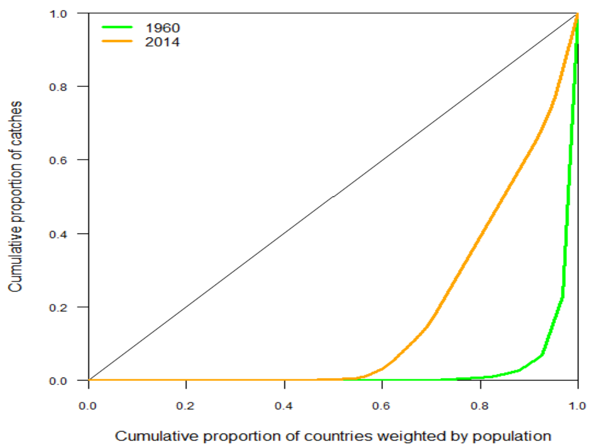


Figure 1.4: Lorenz curves.

The evolution of the Lorenz curves for the distribution of catches shows that countries in the middle and upper parts of the distribution have increased their share in total catches, bringing the curve closer to the diagonal (Figure 1.4). Thus, it can be unambiguously claimed that the 2014 distribution is more equal than the 1960 distribution.

In addition to this graphic result, we are interested in quantifying the inequality in the international distribution of fisheries resources from 1960 to 2014. In particular, we calculated the Gini index, the Atkinson index with parameters 2 (A(2)) and 0.5 (A(0.5)), the Theil index and the Mean Logarithmic Deviation (MLD).

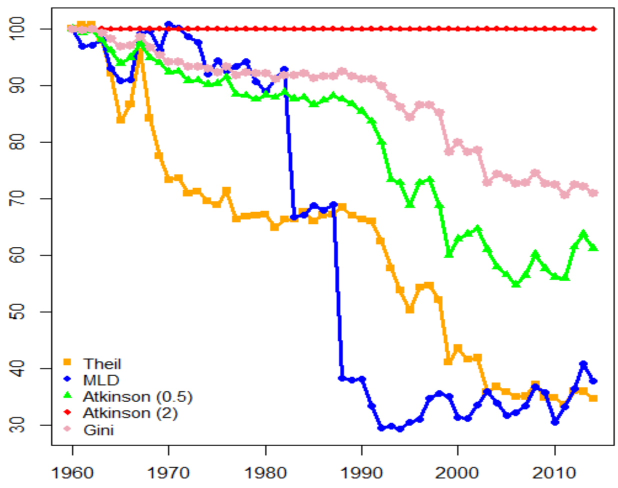


Figure 1.5: Evolution of inequality.

Figure 1.5 illustrates the results. To provide a clearer view of the multiple paths of inequality suggested by the indexes, the inequality values for 1960 are taken as a reference and normalized to 100. Positive and negative fluctuations of inequality place indexes above and below 100, respectively. This enables the magnitude of the changes in inequality to be compared over the period analyzed.

Several features can be highlighted from the evolution of the inequality indexes (Figure 1.5). As previously seen through the evolution of the Lorenz curves, the distribution of catches across countries has become more homogeneous during the period analyzed. In particular, it can be observed that inequality in the distribution of catches has decreased between 29% and 65% from 1960 to 2014, depending on the index. Instead of being monotonic, the evolution of indexes experiences fluctuations from one year to another. Nevertheless, three periods can be clearly distinguished. From 1960 to the mid-1980s, inequality follows a decreasing trend (the largest variation found in the indexes during this period does not represent more than 34% of the initial level). A few years after the recognition of EEZs by the UNCLOS in 1982 (UN, 1982), the decreasing trend in inequality intensifies. This change in the behavior of inequality seems reasonable once the EEZ came into effect, as many fleets operating in the EEZ of foreign countries had to move toward other stocks, in most cases, located in high seas. This was especially the case of the fisheries from the US and Canada and to a considerable lesser extent those fisheries in Northwest Africa (Pauly, 2009). These movements decreased inequality as high seas areas became exploited by more countries. During the last stage, 2000–2014, inequality continues decreasing, but more smoothly. This is due to the stability observed in catches (see Figure 1.1) mainly produced by the boom in aquaculture in this period (Nadarajah and Flaaten, 2017).

Another remarkable feature illustrated in Figure 1.5 is the differences observed between the indexes over time. With the exception of the A(2), all indexes show similar trends. In particular, the trend in A(2) distinguishes for being quite flat. This has also been observed in other studies analyzing distributional issues on environmental resources (Duro, 2012) and is due to the inherent characteristics of the Atkinson family of indexes. A(2) is an index which takes into close consideration the observations located at the bottom of the distribution. This implies that, in our case, the A(2) pays especial attention to countries with zero catches from high seas. Even though their proportion has decreased over the period analyzed, there is still a large amount of countries with zero catches. Thus, changes in the inequality when focusing on the bottom part of the catches distribution are negligible. In this sense, we do not consider that A(2) represents the equity in exploitation of fisheries resources well. The rest of the indexes show similar trends, but the differences among them are relevant from the quantitative viewpoint. The Gini index shows the smallest variation in inequality (apart from the A(2) index) over time, because it is very sensitive to changes in the part of the distribution with most observations, i.e., the low tail of the distribution, in our case. The differences observed in the trend of the Theil index and the MLD also reflect the inherent characteristics of this family of indexes. Both belong to the General Entropy family (Table 1.1), but the Theil index is more sensitive to changes in the upper tail than the MLD. This is why the former quantifies more inequality than the latter for the same distribution.

1.4.5 Decomposition of Inequality by Fishing Areas: Biology vs. Technology

The decomposability property of the Theil index allows calculating how much of the inequality observed in fishery catches can be explained by differences *between* fishing areas and how much by differences across countries operating *within* those areas. The *between* component would represent mostly the biological differences between the fishing areas (species, biodiversity, nutrients availability, temperature, climate conditions, etc.). The *within* component would represent the differences across the fleets fishing within the area. These differences may reflect the technological characteristics of the fleets (fishing gear, vessel size and power, and EEZ boundary), the fishermen capacities or the food preferences of population among countries. Since large dissimilarities in the fleets and fishermen capacity of countries are outlined in (FAO, 2018) and differences in the preferences for fish consumption are difficult to measure and may be overpassed by preferences for exporting fish, we associate this component of the inequality, roughly speaking, with technology.

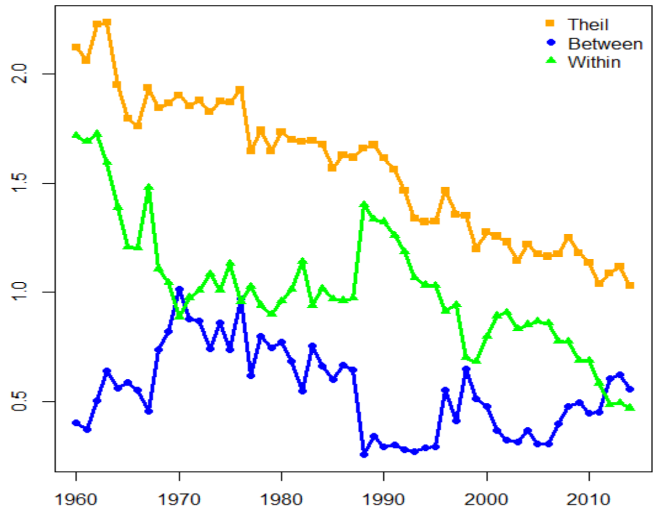


Figure 1.6: *Between* and *within* inequality decomposition of the Theil index.

Figure 1.6 illustrates the evolution of the total inequality and the *between* and *within* components of the inequality in the catches of countries from 1960 to 2014. For an easier comparison of the contributions of each component, these results are not normalized using the initial year as in Figure 1.5. There are two facts worth highlighting in Figure 1.6. On the one hand, we observe that inequality has reduced more than 33% in relative terms between 1960 and 2014. Since the maximum value of the Theil index is given by $Ln(n)$ being in our case $n = 166$, we can say that in 1960 the inequality in the use of fishing resources was about 51% of the maximum inequality while in 2014 the inequality was about 18%.

On the other hand, Figure 1.6 shows that inequality in the high seas catches of countries has been mostly produced by differences *within* fishing areas during the period analyzed. That is, most of the inequality observed is due to the dissimilarity in the average catches between the fleets operating within the same area. This is in line with Sumaila and Teh (2015), who found that only more industrialized countries can access and exploit high seas, being the least developed countries limited to the nearest areas. Nevertheless, the gap between the *between* and the *within* components narrows over time to the point that the *between* component exceeds the *within* component in the last years of the sample. This evolution reflects the technological catching up across countries.

Table 1.4: Proportion of inequality from *between* and *within* fishing areas.

Year	Within	Between
1960	80.89	19.11
1970	46.57	53.43
1980	55.38	44.62
1990	81.94	18.06
2000	62.50	37.50
2010	60.68	39.32
2014	45.71	54.29

Table 1.4 quantifies the Theil decomposition for the key years in percentage terms. As shown in Figure 1.6, the inequality observed seems to have been mainly motivated by inequality among countries fishing *within* the same area (between 45% and 82% of the total inequality is explained by this component). Although its initial contribution seems to maintain at a lower levels during the first decades, the *within* inequality experiences a noticeable increase in the 1990s. This spike might be explained by the implementation of the EEZ in the 1980s. Due to the new legislation, many countries lost their free access to coastal waters previously exploited. In the short term, these countries may have decided to keep their levels of catches by expanding their fishing activities to high seas areas. This sudden movement of the fleets might have result in a quasi-random spread of vessels around global waters, which decreases the biological inequality. At the same time, countries enforced to leave the coastal waters may not have such productive fleets to fish in high seas areas. Consequently, there is more heterogeneity in the fleets of these areas, resulting in the observed increase in *within* inequality. Over the last decades of the period analyzed, the percentage contribution of the *within* inequality has decreased, which reflects the greater reliance of more countries on high seas fisheries and their catching up on fishing technology. Indeed, in the last years, the *within* inequality has overpassed the *between* inequality, which reflects that catches dissimilarities are rather generated by the idiosyncrasies of the areas exploited.

When decomposing the Theil index into *between* and *within* inequality, we can compute the contribution of each area to the latter component. Figure 1.7 shows the contribution of fishing areas to the *within* component. Note that a small (large) contribution by a particular area indicates that the distribution of catches among the countries fishing within this area is quite homogeneous (heterogeneous).

In aggregate terms, Figure 1.7 shows that the Pacific has been responsible for around half of the *within* inequality observed during the whole period. The Atlantic, which generated one third of the initial *within* inequality, presents half of its initial share by the end of the period. On the contrary, the Indian Ocean has

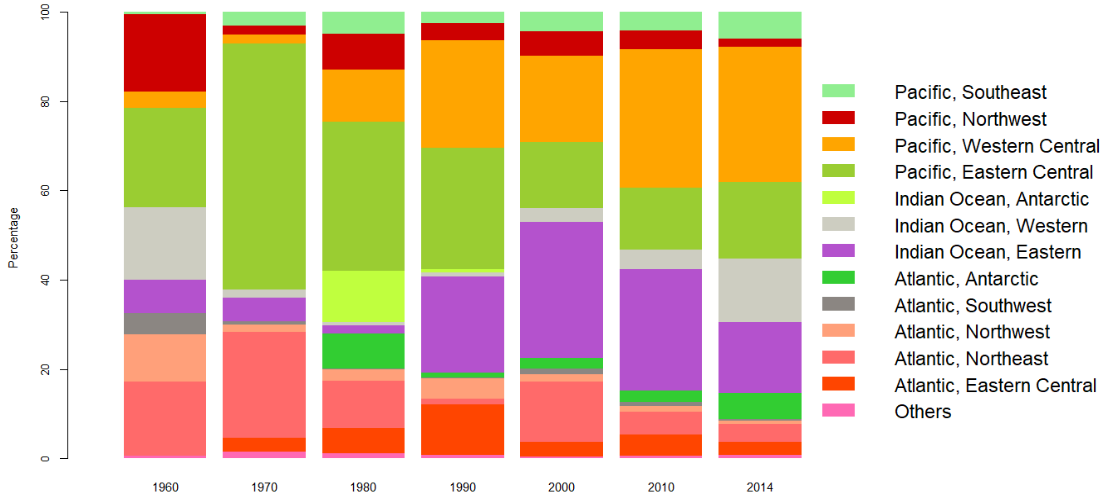


Figure 1.7: Contribution of fishing areas to the *within* inequality. "Others" represents fishing areas whose contribution was below 1% in the years shown.

slightly increased its percentage contribution to *within* inequality during the period analyzed. Even though there is not a single area standing out above the rest, noticeable changes have occurred from 1960 to 2014.

On the one hand, areas such as the Western Central Pacific, the Southeast Pacific, the Eastern Indian Ocean, the Antarctic Atlantic or the Eastern Central Atlantic have increased their percentage contribution to *within* inequality. Within the Western Central Pacific, the concentration of catches seems to have decreased from 1960 to 2014. In the beginning of the period, around 84% of the catches were held by Japan, which represented only 4% of the population of this area. At the end, the largest fishers in this area are Indonesia (with 21% of the catches) and the Republic of Korea (with 20% of the catches), representing 13% and 3% of the population, respectively. Despite this decrease in the percentage concentration, the rise in the catches of this area (Figure 1.3) has motivated such enlargement of the contribution of the *within* contribution. In the Southeast Pacific, Japan represented 85% of the initial catches, with less than 2% of the population. By the end of the period analyzed, the major fishing entity in this area is Ecuador with 50% of the catches and approximately 4% of the population. In this case, it can also be observed that, although there has been a noticeable decrease in the concentration of percentage catches in this area, the increase in the absolute levels of catches (Figure 1.3) results in an increase in the contribution of this area to *within* inequality. Regarding the Eastern Indian Ocean, Japan initially was responsible of all the catches in this area, representing less than 8% of the population. In 2014, Spain becomes the major fishing entity with one third of the catches and 4% of the population of the area. Again, the increase in catches has produced the increased observed in the *within* inequality despite the fall in the concentration of percentage catches. The increase in the *within* contribution of the Antarctic Atlantic is explained by the fact that this area started to be exploited during the period analyzed. Within the Eastern Central Atlantic, there has been also a decrease in the concentration of percentage catches. In particular, the initial major fishing entity was Malta, with 71% of the catches of this area and 0.2% of its population. Ghana constitutes the largest final fishing entity of this area, representing 24% of the catches and 10% of the population in this area. The boost in the absolute catches of this area (Figure 1.3) motivates the increase in its contribution to the *within* inequality.

On the other hand, areas such as the Northwest Pacific, the Southwest Atlantic or the Northern Atlantic have decreased their share in the *within* inequality. In the Northwest Pacific, the decrease in the concentration of percentage catches is noticeable, which may explain the decrease in its contribution to *within* inequality. Particularly, it can be found that this area was initially mostly exploited by Japan (92%), a country which only accounted for 6% of the population. Finally, Indonesia holds the largest amount of catches (32%) while only represents 12% of the population. In the case of the Southwest Atlantic, the concentration of major percentage catches seems to maintain over the period analyzed. Nevertheless, the decrease observed in the catches of this area (Figure 1.3) explains the reduction in the *within* contribution of this area. While the concentration of the percentage catches of the major fishing entities in the Northeast Atlantic decreases, that in the Northwest Atlantic increases. Catches in both of these areas decrease, contributing to the decrease in the *within* inequality in Northeast Atlantic and producing the negative variation in the *within* inequality of the Northwest Atlantic.

1.5 Discussion and Conclusions

While perfectly equal distributions may threaten the target of resources conservation by allowing numerous agents to exploit a limited resource, sustainability policies are also compromised by very unequal distributions, where less benefited agents may not be as prompt to burden with the same costs (Klein et al., 2015; Halpern et al., 2013; Manach et al., 2013). This analysis aims to help policy makers through the illustration of the fishery resources distribution, which may contribute to know the general predisposition of countries to participate in a common conservation plan.

It is well known that fishery catches have traditionally been concentrated in a few countries and fishing areas. This article quantifies the distributional issues that emerge when fisheries exploitation is analyzed over time by applying the inequality metrics used in social science.

Other articles have also used inequality metrics to study different issues arising in fisheries economics. Sumaila et al. (2015) used the Gini index to quantify the distributional effects on profits of the closure of the high seas to fishing. The Gini index was also used by Da Rocha and Sempere (2016) to measure the redistributive effects of restricting the tradability of individual transferable quotas. Bellanger et al. (2016) used the Theil index and its decomposability property to determine the distributional effects of various quota allocation systems among producer organizations. Unlike these articles, our study does not set out to assess the distributional effect of a particular management fisheries policy, but rather to show the unequal use of worldwide fisheries resources over time.

We address the distributional analysis of marine resources as a global common considering catches exclusively from high seas. Catches from the EEZs are not considered a common resource because, from the legal point of view, they are fully under the national jurisdiction of a particular state (UN, 1982). In this context, all inequality indexes confirm that the use of fishing resource is very unequal across countries. However, this use has become more equal over time, with inequality decreasing by between 29% and 65% from 1960 to 2014. This study also shows that, when the geographical origin of the catches is taken into account, until 2000 more than three quarters of the inequality observed in fishing catches is due to technological and fishermen capacity differences across the countries operating *within* fishing areas. This trend is in line with the results of Sumaila et al. (2015), who found that the number of countries fishing in shared areas has doubled from 1950 to 2006. However, this percentage has decreased by more than 35% in the last few decades, reflecting the greater reliance of more countries on high seas fisheries and their catching up on fishing technology. In fact, in the last years of the sample, the *between* component has exceeded the *within* component reflecting the fact that the biological differences between the fishing areas may have widened.

All these findings are very relevant from the policy viewpoint. Policies seeking to ensure the sustainability of marine resources have to take these equity issues into account in the management of the international high seas (Sumaila and Teh, 2015). Jointly with the distribution of fishing effort, the catches of the areas reported in this analysis may provide an idea on whether fishing areas can be further exploited or need to be preserved from their unsustainable exploitation, helping countries to homogenize their catches by redirecting their fishing activity to more productive areas. Nonetheless, an egalitarian use of the fishing resources has to be aligned with other actions that guarantee an equitable use. Other factors such as dependence (i.e., food security), traditional access to the resource or development needs also have to be taken into account for the sustainability of the resource (Schiller et al., 2018; Levine et al., 2015; Belton and Thilsted, 2014). These recommendations are also aligned with the awareness campaigns that promote the sustainability of fisheries by advocating for an eco-labeling of fish that includes information such as the origin of the fish (Kim and Lee, 2018; Bonanomi et al., 2017) or the way in which it has been caught (Onofri et al., 2018).

The search for an egalitarian use of the fishing resources is compatible with the view of the sea as common heritage of mankind rather than a free and open access resource (Schrijver, 2016; Stel, 2016). Some proposals for protecting fishing resources consider the closure of the high seas (Sumaila et al., 2015), the creation of (no-take) marine reserves (Sala and Giakoumi, 2017; Gell and Roberts, 2003; Roberts et al., 2001), or the zoning of the entire oceans, not just the land margin (Russ and Zeller, 2003). Our result are a support for all these proposals as long as the distributional effects are taken into account in the analysis, as Sumaila et al. (2015) did for the case of the high seas closure.

One of the shortcomings of our analysis is that it does not incorporate the management frameworks. Our approach quantifies the inequality of the use of fisheries resource along time but it is not able to explain the reasons of this inequality. High seas are governed by the RFMOs with different management powers to set rules that condition the fishing decisions of the fleets. Even though some of the measures adopted for the RFMOs do not advocate the equal exploitation of the stocks fisheries, this is the case of the European Common Fisheries Policy that allocates quotas among the state members according to the Principle of Relative Stability, which takes into account the historical catch records instead of their population (Hoefnagel et al., 2015; Da-Rocha and Gutiérrez, 2006). In the same line, our approach it is not able *a priori* to take explicitly into account the biological differences of the fishing areas; however, it is able to distinguish *a posteriori* how much of the inequality observed in fishing catches can be explained by differences between fishing areas, which we take mainly as biological differences.

Further research on the availability of fishery substitutes (aquaculture), jointly with an economic valuation of the profits obtained from the commercialization of the catches (including exports), would provide more precise information on the initial willingness of countries to participate in international conservation plans for fisheries. Our analysis skips from the fact that some countries export most of their catches and import other fish products due to the food preference of their people. For instance, Japan exported 16% of fish captured in 2016, according to the FAO data. In contrast, United States exported 31% of fish captured that same year (FAO, 2019a,b). Nonetheless, analyzing the exploitation patterns, instead of consumption patters, allows us to account for that part of the resource that is captured and exported to other countries. We are implicitly assuming that local population benefits from these exports.

From an economic perspective, it would be of interest to quantify the inequality in the distribution of the profits derived from exploitation of the fisheries resources. In Sala et al. (2018), the authors analyzed the profits of several countries obtained very recently from exploitation of high seas fisheries. Some studies (e.g., Pascoe et al. (2019); Rodrigues et al. (2019)) focus on the distribution of profits across agents from specific fisheries but, as far as we know, no studies analyze the distribution of the profits from fishing between countries.

Inequality metrics can be applied to many other distributional concerns of interest in fisheries. One of the shortcomings of our analysis is that catches are taken as a pool without distinguish by species or taxon or by the type of vessel used for harvesting. However, it could be interesting to analyze the distribution of catches by (groups of) species or by types of vessels instead of geographical areas.

Finally, it is worth noting that the catch data used in this study come from the reconstructed series by Pauly and Zeller (2015) in the *Sea Around Us* project (SeaAroundUs). Some studies (e.g., Pauly and Zeller (2017a)) show that data reported to the FAO in recent years by some countries such as China and Myanmar are unreliable because increases in catches may be politically expedient. Moreover, Pauly and Zeller (2017b) highlighted the importance of retroactive corrections in FAO time series to avoid a "presentist bias". Researchers from the FAO refute these criticisms, although they are open to new research that may help to improve their statistical data on fisheries (Ye et al., 2017). In this context of discussion, we have repeated our analysis with the FAO data for the case of the global catches. Results with the FAO data are qualitatively similar to those presented in this article.

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Chapter 2

The role of production factors on landings heterogeneity between EU countries[1]

Distribution of natural resources is considered to be a key aspect in ensuring the success of conservation policies. The Common Fisheries Policy, implemented by the European Union (EU), is an example of long-term international cooperation for sustainability of the marine environment. Nonetheless, continued enforcement of the policy is threatened by its insufficient effectiveness in restoring fish stocks and the tensions that have arisen over unequal distribution of benefits. Recent adoption of the Blue Growth Strategy represents an additional challenge for EU fisheries, since it encourages new alternative economic activities. The present analysis aims to identify ways of enhancing the sustainability of EU fisheries while achieving greater equity in resource distribution and maintaining the activity of the fishing industry, an important staple of many coastal communities in the EU. To this end, the study decomposes heterogeneity amongst per capita landing rates of EU Member States. A number of findings from the decompositions used may be highlighted. Firstly, most of the heterogeneity in per-capita landing rates between Member States occurs within the main EU fishing areas, especially FAO Areas 27 and 37. Secondly, fishing production factors affect per-capita landing heterogeneity to a different extent in Areas 27 and 37. The only exception is the number of fishers, the factor contributing most to heterogeneity in both areas. Technological factors appear to diminish heterogeneity in Area 27 whilst positively contributing to heterogeneity in Area 37. More efficient fleet adjustments could be designed taking into account these contributions by production factors to heterogeneity within fisheries.

2.1 Introduction

Egalitarian international distribution of resources allows global growth to be fostered (IMF, 2017) by providing the poorest countries with incentives to invest in human capital and entrepreneurial activities (Akinci, 2018; Berg et al., 2018); it also foments well-being by reducing the level of poverty and food insecurity as established in the United Nations Sustainable Development Goals (Cisneros-Montemayor et al., 2019; UN, 2015). Furthermore, the distribution of scarce natural resources may be crucial to the success of the international agreements needed

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to ensure their conservation (Drupp et al., 2018; UNDP, 2011; Azar et al., 1996). Ensuring more equitable distribution of resources through the establishment of property rights may threaten sustainability if exploitation of those resources is not properly regulated (White and Costello, 2011; Janmaat, 2005; Scott, 2000). However, very dissimilar exploitation patterns may make it harder for countries to accept the same responsibility for preserving resources (Fabinyi et al., 2015; Hori, 2015; Klein et al., 2015; Halpern et al., 2013).

The development of appropriate ownership schemes has helped to prevent over-exploitation of common-pool resources (Schlager and Ostrom, 1992). In fisheries, distributional concerns and economic inefficiency are linked to inadequacy of property rights (Arnason, 2005; Cochrane, 2000). Consequently, developments in ownership arrangements have become a key ingredient in fisheries management. The 1982 declaration of the Exclusive Economic Zones (EEZs), which recognized the jurisdiction of coastal countries over the natural resources in their 200-nautical-miles adjacent waters (UN, 1982), may be considered as a global system that grants property rights to countries. Creation of the EEZs helped to rebuild certain stocks, especially in countries with science-based fisheries management (Mora et al., 2009) and to protect fisheries from unauthorized fishing (Englander, 2019). From this perspective, EEZs may be viewed as a form of community quota, since only domestic vessels are allowed to access coastal fisheries. However the area to be managed is so large that fisheries authorities have to implement more disaggregated rights-permit schemes, for instance, individual transfer quotas (ITQs) and territorial use rights for fishing (TURFs) programs, which in most cases take into consideration social and equity criteria, in addition to conservation and efficiency benchmarks (Hoshino et al., 2020; Soliman, 2014).

In this context, management of the European Union's fisheries may be considered to be a quite relevant case study since it involves several countries committed to a cooperative approach embodied in the Common Fisheries Policy (CFP). The CFP applies to all EU Member States (MSs), but only 22 of the 27 countries in the Union are coastal. Each coastal state has the right to manage natural resources in its EEZ, but under the CFP, the fishing area of all MSs is considered as a single zone. The main purpose of the CFP is to preserve fishing operations, fish consumption and the marine ecosystems in which EU fleets operate. To this end, the policy has been adapted over time and since the lastest reform in 2013 (EC, 2013), it now covers aspects such as maximizing sustainable yield exploitation for all stocks; reducing discards (the "Landing Obligation"); adapting capital-intensity to fishing opportunities; improving aquaculture to reduce reliance on wild fish and enhancing the role of scientific research in assessing sustainable fishery management. In addition to monitoring the inputs used by countries (such as maximum number of vessels or kilowatts), the CFP establishes fishing *quotas*, i.e. the quantity of fish of each species that can be extracted by each MS, in order to balance fishing operations and fishing opportunities. These quotas are allocated to each country as a fixed percentage of the total allowable catches (TACs). TACs are set by the Council of Fisheries Ministers, based on scientific advice provided by the International Council for the Exploration of the Sea (ICES) and the Scientific, Technical and Economic Committee of Fisheries (STECF). The fixed percentages used to divide the TACs up amongst MSs are determined by the principle of Relative Stability, which gives priority to countries' historical-fishing operations in each fishery (EC, 2019b,a, 2013). In short, the CFP is aligned with the principles of conservation and efficiency of fisheries as well as being committed to the historical fishing rights of Member States.

One of the ways in which MSs circumvent the principle of relative stability is by bringing influence to bear at the annual closed-door negotiations at which TACs are set for stocks of interest to them. Carpenter et al. (2016) found that between 2011 and 2015, the TACs set were on average 20% annually above those proposed in the scientific advice. The MSs benefiting most from this surplus were Denmark and United Kingdom (in absolute terms) and Spain and Portugal (in relative terms). The rigidity of quota regulation has led to the emergence of two instruments enabling different fishing actors (Member States, Production Organizations and fishers) to adapt their fishing operations to their specific needs or preferences. On the one hand, TACs may be landed by foreign operators using domestic vessels under domestic rights, a practice known as "quota hopping" (Hoefnagel et al., 2015). On the other hand, fishing actors may transfer TACs between them ("quota swapping") (Morin, 2000). According to Hoefnagel et al. (2015), 20% of TACs in 2013 were reallocated through international quota swapping.

Taking into account the distributional perspective of the CFP, this article seeks to analyze how landings in EU waters are distributed among MSs. Specifically, the analysis is based on data on the value of landings from 23 MSs between 2008 and 2016 (including the UK as one of the MSs during that period). Although some of the countries considered are characterized by having a set of data-poor indicators (Christou et al., 2019), there are two key advantages to considering this set of countries. Firstly, relevant defining variables for the countries' fleets, such as number of fishers, which are not available in global terms for recent periods, can be entered in the analysis. Secondly, an analysis of the distribution of landings within the EU measured by value shows the particularities of fishing management in the region. In this regard, the distributional analysis was performed by distinguishing the origin of landings between fishing areas, considering those from the Northeast Atlantic, Baltic and North Seas areas (FAO area 27, hereinafter ATW) and those from the Mediterranean and Black Seas (FAO area 37, hereinafter MBS) which represent 62% and 22% of the total value of EU landings, retrospectively, for the period studied. This distinction is relevant, not only because the decline in MBS fisheries contrasts with the improvement observed in trends in ATW fisheries (Vasilakopoulos et al., 2014), but also because the nature of these fisheries is conditioned by differences in the biological characteristics of their ecosystems, the implementation of CFP and their cultural heritage (Smith and Garcia, 2014).

A large body of literature already exists on the international distribution of the use of natural resources and environmental capacity, much of it addressed using analyses of inequality metrics (Bolea et al., 2020; Pozo et al., 2020; He et al., 2019; Chen et al., 2018; White, 2007). This literature has focused to a lesser extent on fishery resources. In particular, the distributional effects generated by specific right-based management systems in fisheries such as ITQs have been assessed by quantifying changes in the distribution of landings and fishing incomes among fishers and boat owners by using inequality metrics (Kourantidou et al., 2021; Villanueva et al., 2019; Villanueva-Poot et al., 2017; Bellanger et al., 2016; Hamon et al., 2009; Baccante, 1995). Another set of articles uses the same approach to analyze the distributional effects of the introduction of ITQs on the industry. In this context, the ownership of catch rights has been concentrated amongst a few large fishers and companies, increasing their market power in, for example, New Zealand commercial fishing (Abayomi and Yandle, 2012) and the Icelandic fisheries (Byrne et al., 2020; Agnarsson et al., 2016). From a global perspective, Gutiérrez and Inguanzo (2019) show that high seas catches are very unevenly distributed among countries; most of the heterogeneity observed is due to dissimilarities in the technological capacity and number of fishers of countries, rather than any biological idiosyncrasies of the fishing areas harvested.

The analysis presented here also follows this international perspective and has a dual objective. On the one hand, it seeks to ascertain to what extent the heterogeneity observed in the distribution of landings (in value per capita) between MSs can be explained by differences between the fishing areas of origin (e.g., species diversity, climate, nutrients, implementation of the CFP and other productivity factors) and to what extent by differences between different fishing actors in these areas (e.g., technological features of the fleet such as gear length, power, and distance and fishers capacities). This issue is addressed by decomposing inequality in landings into its so-called *between-within* components. At the same time, the analysis seeks also to determine the technological reasons for the uneven distribution of landings (in value per capita) between the countries within each fishing area (ATW and MBS). To that end inequality in landings is decomposed into the sum of several components representing the fishing production factors of the countries. Traditionally, catches from fisheries are represented as the result of production factors such as labor and capital services which may also include energy (Da-Rocha et al., 2019). This study focuses mainly on the role of the technological features characterizing fleets. Specifically, it considers the following factor drivers related to different aspects of fishing fleets: technological productivity (measured by landings per kWt of engine power), technical progress (measured by engine power in kWt per vessel), capital-intensity (measured by vessels per fisher) and fishing labor (measured as the percentage of fishers in the total labor force). The breakdown in the inequality indexes set out in Alcalde-Unzu et al. (2009) is used to account for the technological changes observed in the links between the production factors defined.

In view of the characteristics of the EU fleet, it is pertinent to study the role of production factors in determining the distribution of landings among MSs. In 2016, the EU fleet comprised more than 65,000 active vessels, of which 75% were classed as small-scale coastal vessels, 24.6% as large-scale vessels and the remaining remaining 0.4% as distant-water vessels (STECF, 2018). Despite the prevalence of small-scale coastal fisheries (SSCF) in terms of vessels, this segment accounts for just 8% of total gross tonnage and about 30% of engine power (STECF, 2018; Macfadyen et al., 2011). From the perspective of labor input, based on a selection of case studies, Guyader et al. (2013) find that SSCFs in the EU are made up of vessels with smaller crews than larger-scale fleets, although global employment in SSCF amounts to a similar level to that of large-scale fleets. These findings are corroborated in STECF (2018), which quantifies the fishers of SSCF as 51% of the EU fleet and 41% in terms of full-time equivalent units. The relevance of the SSCF also differs between the fishing areas considered; while 22% of the value of catches captured in MBS came from SSCFs, in ATW they represented only 12% of its total value (STECF, 2018). Nevertheless, the effectiveness of the CFP in ensuring the sustainable activity amongst this type of fleet, as required by United Nations Sustainable Development Goal 14.b (UN, 2015) has been questioned (Said et al., 2020; Colla-De-Robertis et al., 2019; Said and Chuenpagdee, 2019; Villasante et al., 2019; Guyader et al., 2013).

Since 2012 Blue Growth has been part of the European Commission Strategy. This plan aims to foster growth in maritime sectors with innovative potential, such as ocean energy, aquaculture, tourism, biotechnology and marine mineral resources, while keeping traditional and new exploitation of the environment within sustainable limits; and both public and private institutions of the MSs are expected to cooperate in the interests of efficient management of marine resources (EC, 2017, 2012). From the perspective of the economic literature, the theoretical and empirical evidences on the relationship between growth and inequality (generally measured in terms of income) are mixed. In general terms, it is accepted that inequality has positive short-term effects on economic performance whereas long-term effects are negative (Berg et al., 2018; Halter et al., 2014). Likewise, heterogeneity in the use of marine resources may, at least in the long term, be expected to affect the blue growth strategy. In this regard, although heterogeneity in landings between countries is not a prominent issue in the blue growth strategy, it is relevant to study its connection with the underlying productivity factors in order to understand and assess the programs in terms of the jobs generated within the EU marine sectors.

Although the per-capita value of landings became more uniform across MS between 2008 and 2016 (STECF, 2019), heterogeneity still causes tension between them. Indeed, it appears to have played a significant role in the

UK's decision to leave the European Union (Agnisola et al., 2019; Forse et al., 2019; O'Higgins and O'Hagan, 2019; Napier, 2016). At the same time, major divergences may be found in the fleets and fishing operations of different MSs (STECF, 2019). The present analysis focuses on two main aspects. The first objective is to detect whether heterogeneity in the per-capita value of landings between MSs is caused by differences in the harvesting area (*between* heterogeneity) or the by dissimilar characteristics of the fishing actors operating in each area (*within* heterogeneity). The second objective is to identify the main production factors leading to heterogeneity in the per-capita value of landings between countries fishing in the same area. The main results show that most of the heterogeneity arises within fishing areas. The number of fishers appears to have been the greatest contributor to per-capita heterogeneity in landings amongst MSs between 2008 and 2016 in the ATW and MBS areas. The effect of technological factors varies within the areas. In the ATW area, technological factors contribute to a decrease in per-capita landing heterogeneity. By contrast, in the MBS area, technological factors augment per-capita landing heterogeneity. Changes in the contribution of technological factors to heterogeneity within each area reflect technological advances in fishing and a reduction in fleet size in Europe in recent years (Lloret et al., 2018; Rousseau et al., 2019).

The remainder of this analysis is structured as follows. Section 2.2 briefly describes the distribution of landings per capita and fishing production factors between MSs and fishing areas. Section 2.3 sets out the *Theil-0* index used to measure heterogeneity and two decomposition used for the analysis: the *between-within* areas decomposition and the multiplicative factor decomposition. The results of the analysis are detailed in Section 2.4. Section 2.5 concludes and discusses the main findings.

2.2 Description of data

Data on landings and the production factors of countries are drawn from the Scientific, Technical and Economic Committee for Fisheries (STECF, 2019). Population data is taken from the World Bank (2019). The data available on fishing activity makes it possible to cover 23 coastal MSs out of a total of 28 EU MSs for the period 2008-2016. The United Kingdom has been included as it was still a MS during that period.

For the purpose of this study, landings are measured in terms of the value of landings, and thus, unless otherwise indicated, any use of the term "landings" henceforth will refer to the value of landings. Since distributional concerns are the main focus, all recorded landings are considered, regardless of their future use (consumption or trade).

2.2.1 Distribution of landings between EU countries

Complete homogenization in the distribution of the value of landings occurs when countries have the same percentages of landings as population. As shown in Figure 2.1, countries' shares in overall landings differ considerably from their respective population shares. These asymmetries appear to have persisted throughout the period analyzed.

In particular, countries such as Denmark, Greece, Ireland, Portugal and Spain have overall landings that exceed their share of the total population, and some have benefited greatly from negotiation of the TACs by the Council of Fisheries Ministers (Carpenter et al., 2016). By contrast, the overall landing shares of Germany, Poland and Romania are lower than their relative population shares. In these countries, the role of commercial

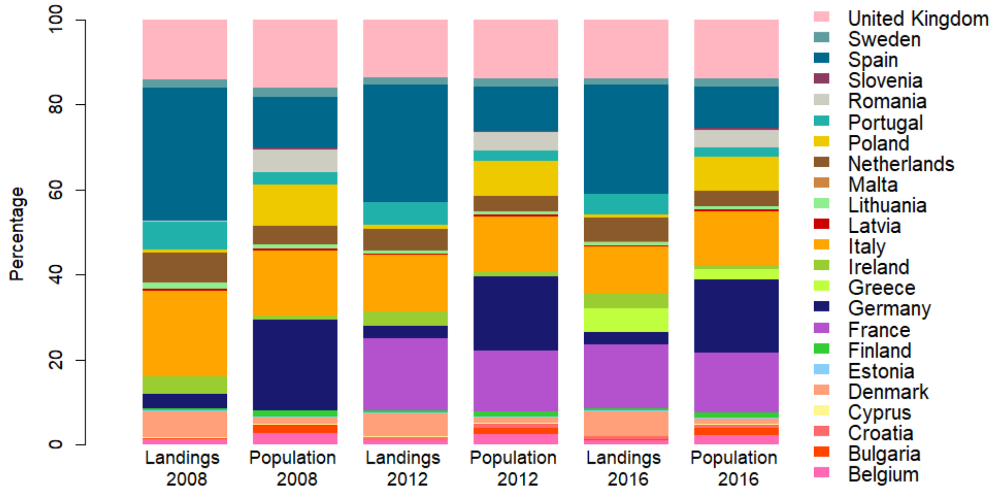


Figure 2.1: Landings value and population of coastal MS (2008, 2012, 2016).

fisheries is small, in economic terms, although inland and recreational fishing and SSCF are increasing in significance (Rakowski et al., 2020; Teodorescu and van den Kommer, 2020; Wedekind et al., 2001).

2.2.2 Distribution of landings between fishing areas

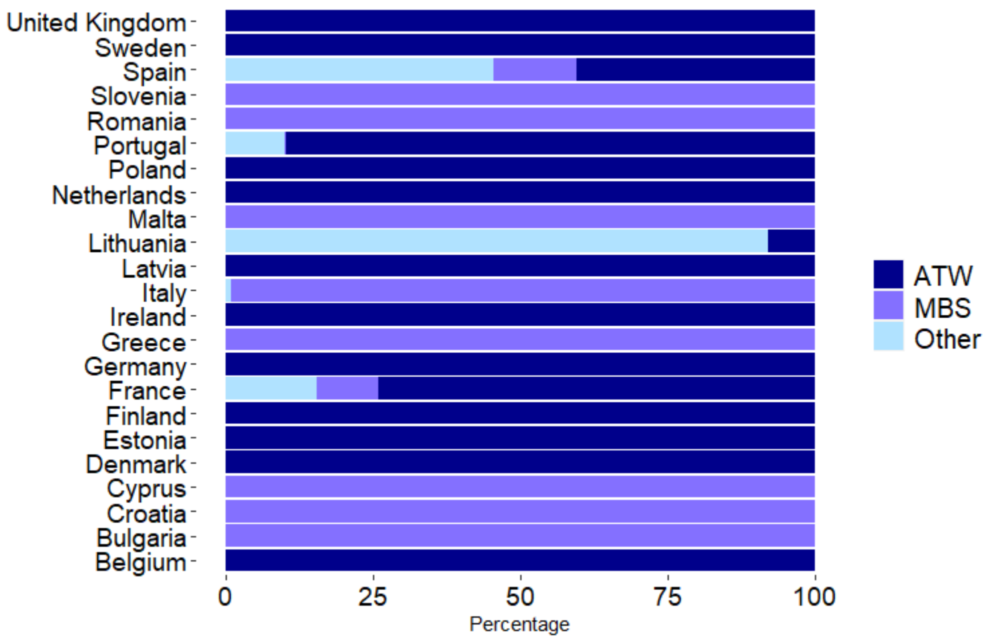


Figure 2.2: Share of the landings value from each area.

The majority of EU fishing activity is concentrated in FAO Areas 27 and 37 (STECF, 2018), which accounted for around 62% and 22% of the value of EU landings between 2008 and 2016, respectively. Nevertheless, their contributions to the value of the landings of MSs differ considerably. In this regard, Figure 2.2 shows a clear distinction between those countries that border Atlantic waters (Northern EU countries, whose income from fishing activity comes from FAO Area 27) and those bordering the Mediterranean and Black Seas (Southern EU countries, whose income from fishing comes from FAO Area 37). Only a few countries bordering both the Atlantic Ocean and the Mediterranean Sea (i.e. France, Portugal and Spain) benefit from harvesting in both of these FAO areas.

Only a limited number of countries within the EU (i.e. Spain, France, Lithuania, Poland, Latvia, Portugal, the Netherlands and Germany) appear to have enjoyed any significant income from fishing activity in other FAO areas (Other Fishing Regions or OFRs) during the period under analysis. Within this group there are two clearly differentiated types of catch: on the one hand, catches from the outermost regions, i.e. the EEZs of the Canary Islands (Spain), Azores and Madeira (Portugal) and the French overseas regions (Guyana, Martinique, Guadalupe, Reunion and Mayotte); and on the other hand, catches from long-distance fisheries in regions outside EU waters (STECF, 2018). The dataset for the present analysis provides complete information for more than 80% of 2016 EU landings from OFR (STECF, 2018). In particular, landings from these areas are considered for Spain, France, Lithuania, Portugal and Italy. See Appendix 2.A for more detailed information on the origin of OFR landings for these countries.

2.2.3 Multiplicative factor decomposition of landings per capita

The IPAT identity describes the multiplicative contribution of population (P), affluence (A) and technology (T) on environmental impact (I) (Commoner, 1972; Ehrlich and Holdren, 1972). Environmental impact may be expressed in terms of resource depletion or accumulation of emissions; population refers to the size of the human population; affluence refers to the level of consumption by that population; and technology refers to the processes used to obtain resources and transform them into useful goods and wastes.

Likewise, the per-capita value of landings for any country is expressed as the result of the interaction of various input factors. In particular, the value of countries' per-capita landings (Impact) can be expressed as the product of the technological productivity and progress of their fleets (Technology), their capital-intensity (Affluence) and the percentage of their population engaged in fishing operations (Population). Mathematically, the value of landings per capita of country i is decomposed as:

$$\frac{Landings_i}{Population_i} = \frac{Landings_i}{Engine\,power_i} \times \frac{Engine\,power_i}{Vessels_i} \times \frac{Vessels_i}{Fishers_i} \times \frac{Fishers_i}{Population_i}, \tag{2.1}$$

where *Engine power* refers to total kilowatts (KWt), *Vessels* to the number of units used in commercial fishing and *Fishers* to the number of fishing workers; all in reference to the fleet of country i.

The $Landings_i$ / $Engine\ power_i$ ratio denotes the productivity of the aggregated fleet in country i. In fisheries, this is usually referred to as LPUE (landings per unit of effort) (STECF, 2018). Fishing effort can be measured either by the natural characteristics of fishing vessels (such as engine power) or by fishing operations (for example, number of days fishing) (Zhou et al., 2003). This ratio represents the productivity of fleets considering the first of these criteria.

The $Engine\ power_i$ / $Vessels_i$ ratio reflects the average engine power of vessels in country i. A positive change in this indicator means that on average, vessels become more powerful; this ratio is therefore associated with the technological level of the fleet of country i.

The $Vessels_i$ / $Fishers_i$ ratio measures the relationship between physical capital and labor for the aggregated fleet of country i. Higher values for this ratio indicate that fleets are more capital-intensive, a factor that is associated with smaller scale fleets as more vessels are employed per fisher. This is a characteristic of less industrial fleets. By contrast, lower values of this ratio are associated with larger scale fleets since fewer vessels are used per fisher.

The last ratio, $Fishers_i$ / $Population_i$, shows the scale of the fishing sector in the labor force of economy i.

To simplify the comments on analyses and their results, the above terms are referred as technological productivity (Landings per Kilowatts, LPK), technical progress (Kilowatts per Vessel, KPV), capital-intensity (Vessels per fisher, VPF) and labor participation (Fishers per population, FPP). Applying this notation, the decomposition can be rewritten as:

$$LPC_i = LPK_i \cdot KPV_i \cdot VPF_i \cdot FPP_i.$$

The countries in the sample show great diversity in the production factors defined above (LPK, KPV, VPF and FPP). Figure 2.3 shows the average use of these production factors by MS. In particular, the distribution of the factors is shown in three scenarios: the ATW scenario, with the distribution of factors engaged exclusively with fishing in FAO area 27; the MBS scenario, with the distribution of factors engaged exclusively with fishing in FAO area 37; and the EU scenario with the overall distribution of factors.

The distribution of technological productivity shows that average returns from fishing differed significantly between countries in the period analyzed, ranging from almost 73 to around 2060 Euros per kilowatt (Figure 2.3(a)). The subtle division between Western and Eastern EU countries in the general framework becomes noticeable when we examine the ATW and MBS areas separately (Figures 2.3(b) and 2.3(c)). Here, Western EU countries can clearly be seen to enjoy a much higher average return than Eastern EU countries.

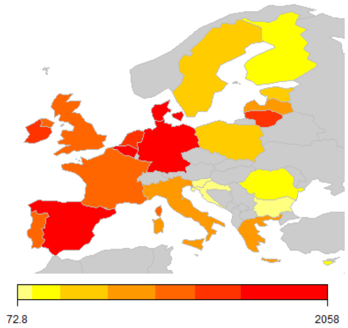
Central EU countries appear to have employed more high-powered vessels in fishing than other EU countries during the period under analysis (Figure 2.3(d)), especially in the ATW area (Figure 2.3(e)). In the MBS area (Figure 2.3(f)), average use of technical progress is a differentiating characteristic of Western EU countries, which use more high-powered vessels.

The distribution of capital-intensity shows that Western EU countries use more industrial fleets than most of Eastern EU countries (Figures 2.3(g), 2.3(h) and 2.3(i)). As for the respective areas, the use of industrial fleets is greater in MBS area than in ATW area. In particular, the average number of fishers by vessel in MBS ranges between 1 and 12 whereas in ATW area ranges between 1 and 5.

Compared to other regions, Europe has one of the lowest percentages of overall population working in the fishing industry (FAO, 2018). Within the EU, the percentage of the population employed in the sector varies markedly from country to country (Figures 2.3(j), 2.3(k) and 2.3(l)). In general, the largest ratios of fishers-per-capita are found in the Southern countries.

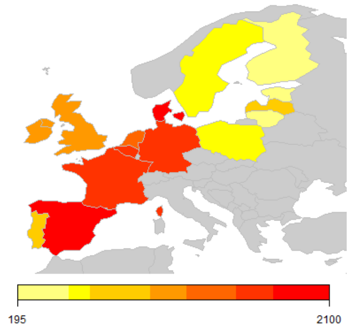
Figure 2.4 shows the use of resources by each country in the ATW and MBS areas. Countries are classed into three groups depending on the average use of the corresponding production factor in each area between 2008 and 2016. If its average use of a certain factor is in the lowest third of the distribution, the country is assigned to the First tercile (1^{st} tercile) category. A country is placed in the Second tercile (2^{nd} tercile) category when it is in the middle third of the distribution. Finally, the country is placed in the Third tercile (3^{rd} tercile) category if it is in the highest third of the distribution.

Given that no common pattern can be drawn, rather than focusing on each country, details are given of those with the highest and lowest landings per capita. Denmark, which is one of the countries with the highest rate of landings per capita, harvests only in the ATW area. Within this area, Denmark stands in the highest third of the distribution for technological productivity, technical progress and capital-intensity. The only exception is the ratio of fishers per capita, where it stands in the medium range. This may be explained by the major role of industrial vessels in Denmark, which is characterized by being one of the largest producers of fishmeal and fish oil in Europe. In 2016, 97% of landings were caught by semi-industrial or industrial vessels, representing



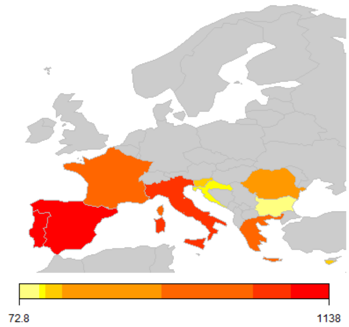
(a) Technological Productivity

(Landings per Kilowatt, LPK), EU



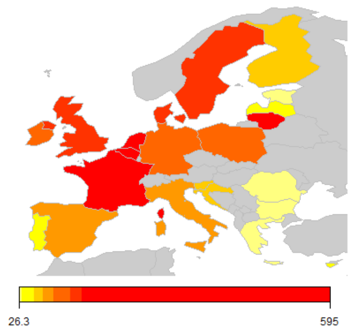
(b) Technological Productivity

(Landings per Kilowatt, LPK), ATW



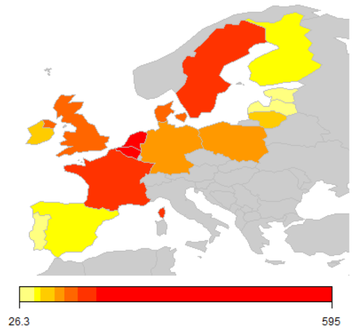
(c) Technological Productivity

(Landings per Kilowatt, LPK), MBS



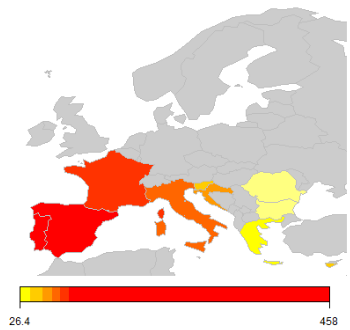
(d) Technical Progress

(Kilowatts per vessel, KPV), EU



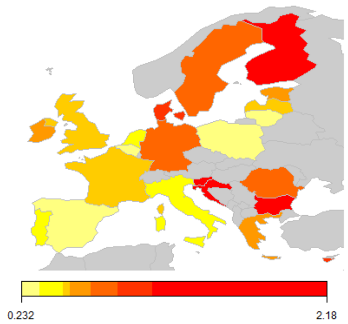
(e) Technical Progress

(Kilowatts per vessel, KPV), ATW



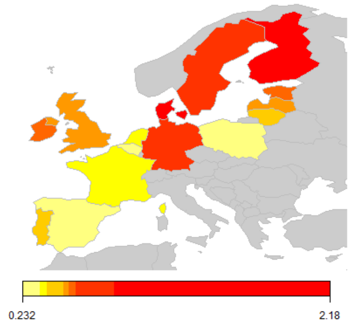
(f) Technical Progress

(Kilowatts per vessel, KPV), MBS



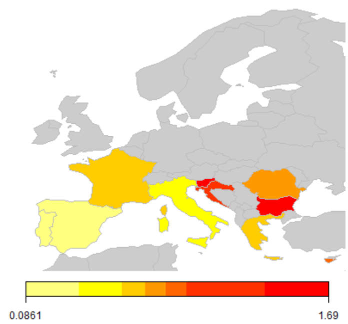
(g) Capital-intensity

(Vessels per fisher, VPF), EU



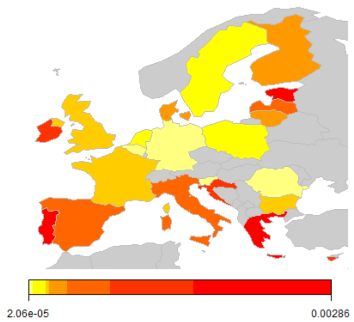
(h) Capital-intensity

(Vessels per fisher, VPF), ATW



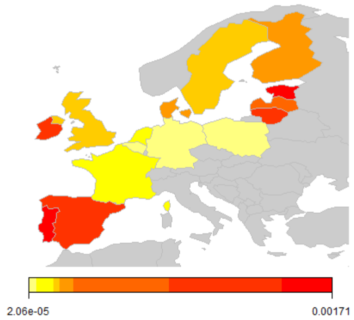
(i) Capital-intensity

(Vessels per fisher, VPF), MBS



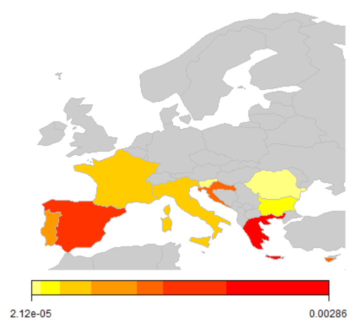
(j) Labor participation

(Fishers per population, FPP), EU



(k) Labor participation

(Fishers per population, FPP), ATW



(l) Labor participation

(Fishers per population, FPP), MBS

Figure 2.3: Distribution of fishing production factors across coastal MS. Average from 2008 to 2016. EU: all areas; ATW: FAO area 27; MBS: FAO area 37. Values increase with color intensity, from yellow to orange and red.

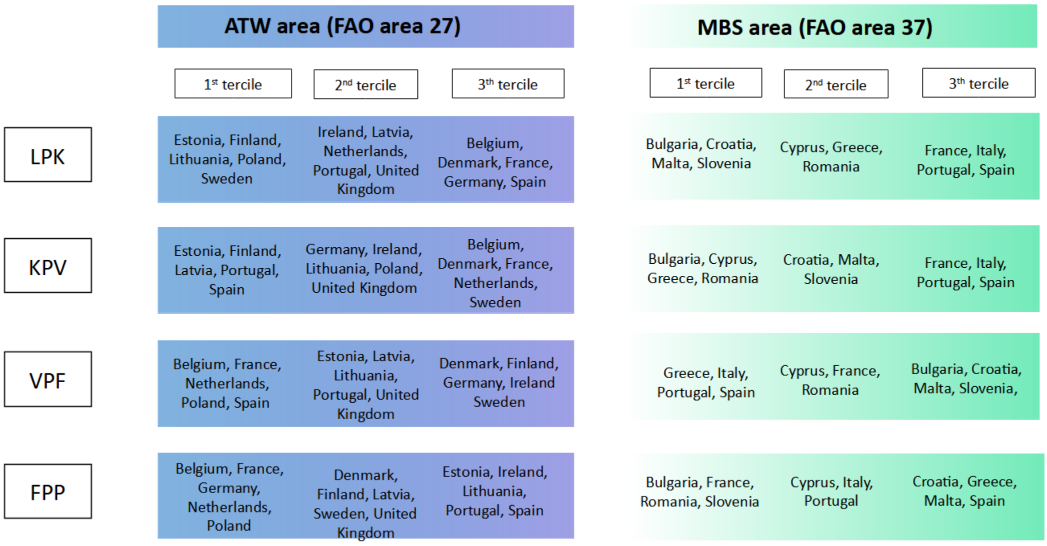


Figure 2.4: Relevance of production factors by fishing areas and countries. LPK (Landings per Kilowatt), KPV (Kilowatts per vessel), VPF (Vessels per fisher), FPP (Fishers per population).

around 25% of active vessels in the Danish fishing fleet (Carvalho et al., 2020). Ireland is another example of a country fishing only in the ATW area with a large ratio of landings per capita. Ireland's technological productivity and technical progress lie in the middle third, but in terms of capital-intensity and fishing labor per capita it is in the top third of distribution. Greece is also among the countries with the highest ratio of landings per capita. However, the country's fishing activity is limited to the MBS area. Within this area, Greece is in the lowest third of technical progress and capital-intensity, in the middle third for technological productivity and in the highest third for fishing labor per capita. Portugal, one of the countries with the highest landings-per-capita ratios, has fishing activity in both the ATW and MBS areas. However, its profile varies from one area to another. In the ATW area, Portugal is in the highest third of fishing labor per capita, in the middle third of technological productivity and capital-intensity and in the lowest third in technical progress. In the MBS area, Portugal is in the highest third of capital productivity and technical progress, the middle third of capital-intensity and the lowest third of fishing labor per capita. Like Portugal, Spain is among the countries with the highest levels of landings per capita in the ATW and MBS areas. However, there is less disparity in its profile in the two areas. Spain is in the highest third of technological productivity and fishing labor per capita and in the lowest third of capital-intensity. In technical progress, the country is in the highest third in the MBS area, but in the lowest third in the ATW area. Germany has one of the lowest landings-per-capita levels of all countries. It fishes in the ATW area and its fleet is characterized by high levels of technological productivity and capital-intensity, medium levels of technical progress and low levels of fishing labor per capita. Poland, which is similar to Germany in terms of low landings per capita and fishing activity location, belongs to the lowest third of technological productivity, capital-intensity and fishing labor per capita. Romania is also among the countries with low landings per capita. In contrast to Germany and Poland, Romania's fishing activity is confined to the MBS area. Within this area, Romania is in the medium third of technological productivity and capital-intensity and in the lowest third in technical progress and fishing labor per capita.

2.3 Methodology

2.3.1 Measuring heterogeneity

The inequality metrics approach has been used to quantify heterogeneity in the distribution of the variables of interest. In particular, this study measures heterogeneity using the second measure of Theil proposed by (Theil, 1967), also called the Mean Logarithmic Deviation index. This index can be derived from the Generalized Entropy family (Shorrocks, 1984, 1980) for a parameter value equal to zero ($\alpha = 0$).

For the purpose of this study, this inequality measure is referred to as the *Theil-0* index, denoted by T, and its application to the distribution of landings per capita (LPC) can be expressed as:

$$T(LPC) = \sum_{i}^{n} p_i \cdot \ln\left(\frac{\overline{LPC}}{LPC_i}\right), \tag{2.2}$$

where the subindex i refers to each of the n countries, p_i weights the observations of countries according to their share of the total population and $\overline{LPC}$ refers to the overall average for landings per capita. The greater the value of the index, the greater the disparities between the fishing areas considered. In the extreme case where landings per capita of countries are exactly the same, this index takes a value of $T = 0$. Thus, values closer to zero reflect more even distributions of landings per capita.

The *Theil-0* index satisfies the basic properties of anonymity, population and scale invariance and the Pigou–Dalton transfer principle. Moreover, apart from the absolute Gini index, this index is the only measure that respects both the principle of transfer and the principle of monotonicity in distance, for which reason Shorrocks (1980) argued that this index is the *"most satisfactory of the decomposable measures"* because it unequivocally decomposes overall inequality into the contribution from the inequality within subgroups and that from the inequality between subgroups, for any partition of the population.

2.3.2 Decomposition of heterogeneity by fishing areas

One of the advantages of using the *Theil-0* index is that it can be usefully decomposed, in order to evaluate the impact of the *between-within* components whenever the population can be partitioned into exclusive subgroups. This is the case of this study where the population (landings) are available at country level and can be sorted by fishing area of origin. Given this property, it is possible to quantify how much of the observed heterogeneity in the distribution of the landings can be explained by differences between the fishing areas of origin and how much by differences between the fishing actors operating within these areas. Figure 2.5 represents the logic behind this decomposition.

Formally, for the application of this study, the *Theil-0* index can be decomposed as:

$$T(LPC) = T_B + T_W,$$

being

$$T_B = \sum_{g} p_g \cdot \ln\left(\frac{\overline{LPC}}{\overline{LPC}_g}\right),$$

$$T_W = \sum_{g} p_g \cdot T(LPC_g),$$

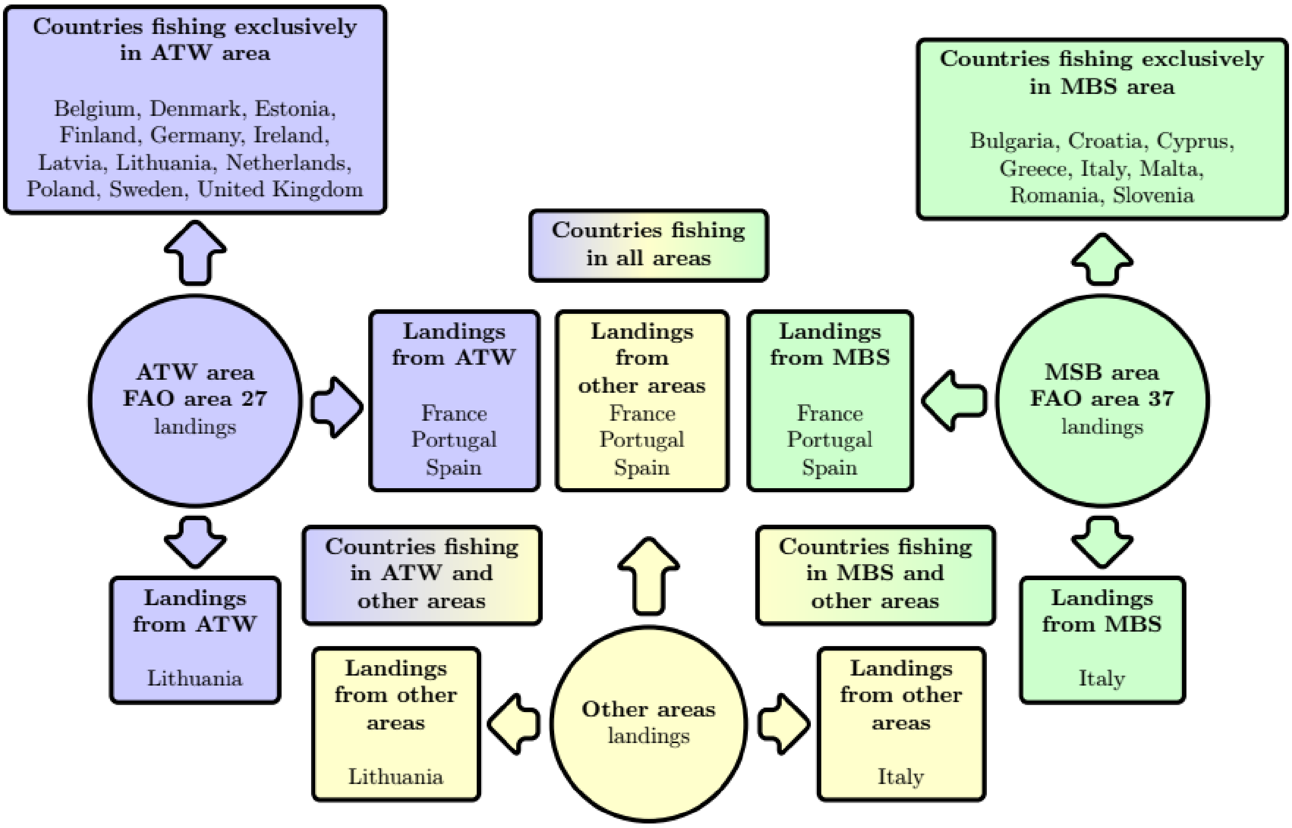


Figure 2.5: The logic behind the *between-within* heterogeneity decomposition. Landings of every country can be assigned to one of the fishing areas. The between component represents the heterogeneity of landings between the fishing areas (between countries of different colors) and the within component represents the heterogeneity of landings within the areas (within countries with the same color).

where p_g is the proportion of population attributed to the area g, $\overline{LPC}_g$ is the average for landings per capita in area g, and $T(LPC_g)$ is the *Theil-0* index calculated considering exclusively the landings in area g, being $g = \{ATW, MSB, Others\}$ (more details in Appendix 2.B). For the purposes of allocating the population of each country to fishing areas an equal distribution principle is assumed (see more details in Gutiérrez and Inguanzo (2019)).

Note that the weights in the within-subgroup add up to one and do not depend on the mean of per capita landings for the area. This characteristic has been referred to as path-independent decomposability (Foster and Shneyerov, 2000) and means that the additive decomposition of the index is independent of the path followed to define the two components. The above characterization shows that the overall heterogeneity is the sum of the weighted sum of heterogeneity within areas and the heterogeneity between areas.

The terms T_W and T_B are the within and between components, respectively. The within component accounts for heterogeneity inside each area and the between component accounts for heterogeneity between areas. In this study, landings are available at country level and can be classified by fishing area of origin. Given that the fishing areas considered (ATW, MBS and the remainder) are so dissimilar, it is of interest to ascertain what proportion of the heterogeneity observed is due to differences within and between fishing areas.

2.3.3 Factor decomposition in fishing areas

When the value of per-capita landings is decomposed by input factors, as proposed in Section 2.2.3, it is of interest to study to what extent each factor contributes to heterogeneity in the distribution of landings between countries.

If the production factors in the decomposition were independent of one other, the heterogeneity of landings per capita estimated by the *Theil-0* index (T(LPC)) would be equal to the sum of the *Theil-0* index applied to the four production factors (see Appendix 2.C), i.e.

$$T(LPC) = T(LPK) + T(KPV) + T(VPF) + T(FPP). \tag{2.3}$$

Table 2.1: Pearson correlation between production factors.

	Area 27				Area 37			
	LPK	KPV	VPF	FPP	LPK	KPV	VPF	FPP
LPK	1.00	0.32	-0.38	-0.23	1.00	0.58	-0.73	-0.05
KPV	0.32	1.00	-0.31	-0.46	0.58	1.00	-0.54	0.06
VPF	-0.38	-0.31	1.00	-0.16	-0.73	-0.54	1.00	-0.25
FPP	-0.23	-0.46	-0.16	1.00	-0.05	0.06	-0.25	1.00

LPK (Landings per kilowatt), KPV (Kilowatts per vessel), VPF (Vessels per fisher), FPP (Fishers per population).

However, the factors are dependent by construction. Table 2.1 shows the empirical Pearson correlations between the factors for the two main fishing areas considered. Almost all have a negative relationship. The only common exception between areas is the link between technological productivity and technical progress, which may indicate that more technologically advanced vessels have greater capacities and can seek more productive areas further away (Rousseau et al., 2019; Lloret et al., 2018). The large magnitude of the negative relationship between technical progress and capital-intensity and between technical progress and fishing labor between areas reinforces the idea that two kinds of fleet coexists (Rousseau et al., 2019): one more artisanal (using more capital and labor, but with less technical progress) and the other more technologically oriented (less numerous, but with more technical progress). In the MBS area, the correlation between technological productivity and capital-intensity is also notable, suggesting that increases (decreases) in technological productivity are associated with decreases (increases) in capital-intensity. Consistently with STECF (2018) and Lloret et al. (2018), this shows that industrial vessels (with larger crews per vessel) are associated with larger fishing returns. The relationship between capital-intensity and fishing labor per capita reflects the substitutability of these two factors.

Thus, in order to analyze the importance of each production factor in the heterogeneity of landings distribution it is necessary to take into account the interrelationships between the factors. Moreover, given the differences between correlations in the two areas the factor decomposition for the areas is expected to be different. To that end, the Theil decomposition proposed by Alcalde-Unzu et al. (2009) was applied to the data set for the ATW and MBS areas. Broadly speaking, this procedure decomposes the *Theil-0* index as the sum of the index for each factor considered plus an additional element reflecting the interrelations between the factors. Formally, the *Theil-0* index associated with the heterogeneity of landings per capita can be calculated as,

$$T(LPC) = T(LPK) + T(KPV) + T(VPF) + T(FPP) + \ln\left(\frac{\overline{LPC}}{\overline{LPK}\cdot\overline{KPV}\cdot\overline{VPF}\cdot\overline{FPP}}\right), \tag{2.4}$$

where the overline symbol on a variable reflects the weighted average of the corresponding variable for all countries. Each of the first four summands reflects the direct impact of each factor on landings heterogeneity and the fifth summand represents the indirect impact of all factors together due to their interrelationship. Alcalde-Unzu et al. (2009) show that this element can be expressed in terms of covariances between factors. See Appendix 2.C for a full characterization.

2.4 Results

2.4.1 Heterogeneity in fishing areas

Table 2.2 shows the *between-within* decomposition of heterogeneity in landings-per-capita among coastal MSs grouped by fishing areas (ATW, MBS and Others) and evolution of these figures from 2008 to 2016. The results show that landings per capita are more heterogeneously distributed within fishing areas than between them. This implies that ecological idiosyncrasies of fishing areas such as species composition of fish stocks, biodiversity, nutrients availability, temperature, climate conditions, etc. (underlying the between component) play a minor role in landings per capita heterogeneity amongst MSs when compared to the effects of using dissimilar fleets by countries harvesting in the same area (producing the within component).

Heterogeneity in the landings per capita appears to have decreased by around 27% from 2008 to 2016, mainly due to the observed decrease in heterogeneity within fishing areas. This suggests that landings per capita between MSs have become more alike due to the homogenization of their fleets, which may be driven by the reduction in fleet capacity projected in the CFP (EC, 2013) as well as improvements in fishing technologies (Rousseau et al., 2019). Despite the decrease over time, heterogeneity within fishing areas represents more than 90% of the total heterogeneity in landings per capita from 2008 to 2016.

The contribution of each area to the within heterogeneity can also be observed to be significantly different (second block in Table 2.2). In particular, the ATW area accounts for more than 60% of total within hetero-

Table 2.2: Distribution of landings per capita among countries and fishing areas. Heterogeneity decomposition in *between-within* components.

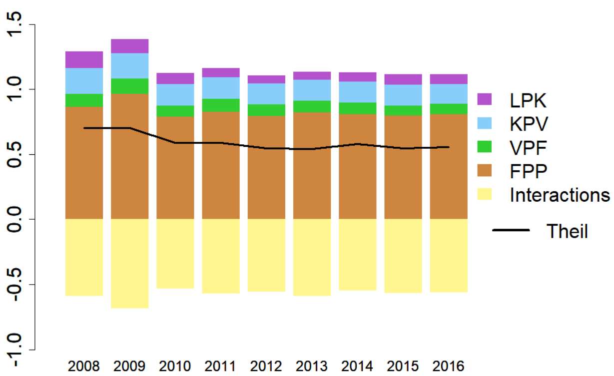
	Theil	Decomposition				Contribution to the within heterogeneity					
		Between		Within		ATW area		MBS area		Other areas	
Year	Abs.	Abs.	%	Abs.	%	Abs.	%	Abs.	%	Abs.	%
2008	0.86	0.04	4.31	0.82	95.69	0.49	59.11	0.34	40.78	0.00	0.11
2009	0.89	0.05	5.68	0.84	94.32	0.49	58.27	0.35	41.66	0.00	0.08
2010	0.75	0.03	3.73	0.72	96.27	0.41	57.36	0.30	41.91	0.01	0.74
2011	0.69	0.03	4.17	0.66	95.83	0.41	61.37	0.25	37.97	0.00	0.66
2012	0.68	0.04	5.40	0.64	94.60	0.37	58.05	0.26	40.88	0.01	1.08
2013	0.66	0.05	6.96	0.62	93.04	0.37	60.07	0.24	39.05	0.01	0.88
2014	0.65	0.05	7.08	0.61	92.92	0.38	62.97	0.22	36.13	0.01	0.90
2015	0.59	0.03	5.45	0.56	94.55	0.37	66.44	0.18	32.89	0.00	0.66
2016	0.63	0.03	5.08	0.60	94.92	0.38	63.10	0.21	36.01	0.01	0.89

geneity. The MBS area represents around 40% of total within heterogeneity while the contribution of Other areas remains below 1% throughout most of the period analyzed. Between 2008 and 2016, heterogeneity within ATW and MBS areas decreased. Nevertheless, the previous pattern was maintained during the period under analysis.

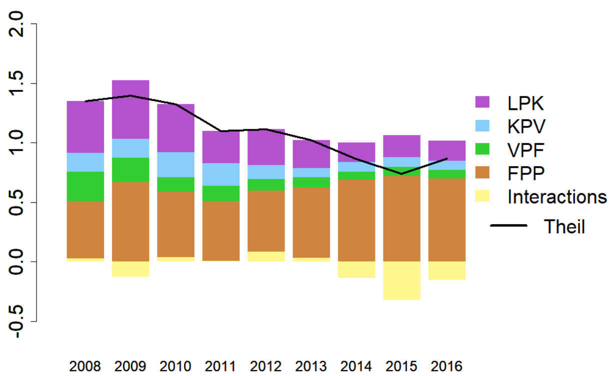
The major role played by fleet dissimilarities in explaining landings-per-capita heterogeneity between MSs justifies further exploration of the production factors leading to such heterogeneity.

2.4.2 Factor decomposition in fishing areas

Given that ATW and MBS fishing areas represent most of the landings-per-capita heterogeneity, this Section focuses on the factor decomposition of the heterogeneity found in these areas.



(a) ATW area



(b) MBS area

Figure 2.6: Decomposition of landings per capita heterogeneity by input factors: Technological productivity (Landings per Kilowatt, LPK), technical progress (Kilowatts per Vessel, KPV), capital-intensity (Vessels per fisher, VPF) and labor participation (Fishers per population, FPP).

Figure 2.6 illustrates the decomposition of the landings-per-capita heterogeneity for each fishing area. The factor decompositions reveal that the reasons for the landings-per-capita heterogeneity vary between areas. The only similarity found in the decomposition of heterogeneity in the two areas is the major role played by fishing labor per capita. Indeed, the dissimilarities in the number of fishers per capita between MSs is the reason for most of the heterogeneity found in their landings per capita, regardless of the area harvested. Heterogeneity in the technical progress of fleets is the second major reason for landings-per-capita heterogeneity in the ATW area. However, technological productivity makes the second largest contribution to landings-per-capita heterogeneity in MBS area. Similarly, the interactions between production factors affect the heterogeneity in each area differently. Thus, interactions significantly reduce landings-per-capita heterogeneity in the ATW area, while having scarcely any effect on landings-per-capita heterogeneity in the MBS area. The overall heterogeneity in landings per capita of each area can be obtained by adding the contributions of production factors and their interactions. A comparison of total heterogeneity shows that landings per capita are more unevenly distributed between countries in the MBS area.

Table 2.3: Percentage contribution of input factors to heterogeneity in landings per capita.

	ATW area				MBS area			
Year	LPK	KPV	VPF	FPP	LPK	KPV	VPF	FPP
2008	-2.41	7.24	-6.77	101.93	32.67	12.24	18.67	36.42
2009	-8.91	3.42	-7.40	112.89	32.70	9.30	12.38	45.62
2010	-8.17	5.24	-7.73	110.65	31.20	16.77	10.05	41.98
2011	-12.47	3.79	-7.81	116.48	24.89	17.65	11.96	45.50
2012	-14.92	3.93	-9.10	120.09	28.89	12.86	10.36	47.89
2013	-16.15	2.88	-10.87	124.14	23.51	8.59	9.13	58.77
2014	-11.88	4.54	-8.88	116.22	14.67	5.49	3.88	75.96
2015	-11.92	3.65	-11.75	120.02	14.10	-0.04	-0.33	86.28
2016	-11.69	2.11	-10.20	119.79	15.24	4.76	3.80	76.20

LPK (Landings per kilowatt), KPV (Kilowatts per vessel), VPF (Vessels per fisher), FPP (Fishers per population).

The direct and indirect effect of each factor on overall heterogeneity of landings per capita cannot be quantified in any straightforward way given the dependency between the production factors. Since there is no single way of distributing the indirect impact between factors (Shorrocks, 1982), it is frequently distributed equally amongst them (Jusot et al., 2013). Following this procedure, Table 2.3 shows the percentage contribution of each production factor to the heterogeneity in landings per capita in the ATW and MBS areas over the period analyzed.

The heterogeneity in the number of fishers per capita generates most of the heterogeneity in the per-capita landings between MSs. The impact of this factor on heterogeneity of landings rose considerably from 2008 to 2016. In particular, the contribution of fishing labor per capita to landing heterogeneity increased by 18 and 40 percentage points in ATW and MBS, respectively. Technical progress also affects positively landings heterogeneity in both areas. However, its contribution decreased over the period analyzed. The contribution of technological productivity and capital-intensity to landings heterogeneity is entirely different in each area. In

ATW, these factors contributed negatively to landings-per-capita heterogeneity during the period analyzed. By contrast, these factors increased landing heterogeneity in the MBS area, especially technological productivity. The evolution of technological productivity, technical progress and capital-intensity led to the decrease observed in landings-per-capita heterogeneity in both areas from 2008 to 2016. This homogenization in technological factors reflects both the advances in fishing technology of MSs (Rousseau et al., 2019) and the limitation of the fleet capacity by the CFP (EC, 2013).

2.5 Discussion and conclusions

The international distribution of resources has transcended the normative role traditionally assigned to it (Bennett, 2018) and is recognized as a path for expanding global economic growth and enhancing well-being (Akinci, 2018; Berg et al., 2018; IMF, 2017; UN, 2015; Alesina and Perotti, 1996). With regard to the international distribution of natural resources, the consequences of their allocation additionally extend to the future availability of those resources (Rice, 2021; Hori, 2015; Klein et al., 2015) and among these natural resources, fisheries are no exception (Österblom et al., 2020).

Fishery resources represent more than half of global fish production and are mostly used for human consumption. Fishing has very significant socioeconomic effects, given that as at least 39 million people are estimated to be engaged in the industry (FAO, 2020). Because the proportion of fish stocks exploited in excess of sustainable limits keeps increasing, international agreements need urgently to be implemented if fisheries are to be conserved (FAO, 2020; UN, 2015). However, inequity in the distribution of fishery resources undermines the cooperation required for the conservation of the marine environment and in numerous cases at a global level leads to serious conflict (Kourantidou et al., 2021; Llompart et al., 2017; Penney et al., 2017; McClanahan et al., 2015; DuBois and Zografos, 2012).

The CFP implemented by the EU in the 1970s is an example of the international agreements that are required to target sustainable exploitation of fisheries. This framework sought to protect EU fisheries against overexploitation by controlling the fishing activity of MSs whilst also allowing them to obtain the best possible socioeconomic returns. Apart from the objective of decreasing overall fleet capacity, additional controls were established for each fishing area (EC, 2019b, 2013). EU fishing activity is mainly concentrated in the ATW and MBS areas, which accounted for more than 80% of the value of EU landings between 2008 and 2016. Only a few MSs (i.e. Spain, France, Lithuania, Poland, Latvia, Portugal, Netherlands and Germany) fished in Other areas during this period (STECF, 2019). In ATW, fishing activity is mainly regulated through Total Allowable Catches (TACs), which specify the amount of each species that can be landed (EC, 2019a). These TACs are set annually and split into quotas amongst MSs safeguarding the Principle of Relative Stability (PRS). The PRS prioritizes the fishing activity of countries that originally harvested a given area, in order to protect communities historically dependent on those resources (EC, 2019b, 2013; Da-Rocha and Gutiérrez, 2006). By contrast, the most common regulations in the MBS area are the technical measures limiting certain fishing practices as well as the size, number and selectivity of fishing gears (EC, 2006). In general, the CFP appears to have succeed in reducing the EU's fleet capacity. Between 2008 and 2018, the number of vessels fell by 5.5%. Simultaneously, the total power (kW) and tonnage (GT) of the EU fishing fleet decreased by almost 17 and 10.5 percent points respectively (STECF, 2020). Nevertheless, the CFP has been called into question on multiple fronts. From the perspective of efficacy, the CFP guidelines have not been enough to safeguard the sustainability of all EU fish stocks (Froese et al., 2018; Cardinale et al., 2017; Da-Rocha et al., 2012; Villasante et al., 2011). From an institutional viewpoint, lack of transparency in definition of the regulation is an obstacle to stakeholder-approval and engagement (Garza-Gil et al., 2017; Khalilian et al., 2010). Indeed, it has been claimed that asymmetries in the bargaining power of different countries and interest groups have been the main reason for inconsistencies in the distribution of CFP benefits (Orach et al., 2017; Carpenter et al., 2016). In this sense, the undervaluation of small-scale fisheries, which represents 75% of the active EU fleet and half of the EU fishing labor force (STECF, 2020), underlies the heterogeneous impact of different groups on the distribution of benefits in the CFP and the CFP's compliance with the goal of sustainable development (Said et al., 2020; Said and Chuenpagdee, 2019). From an ecological perspective, variations in the distribution of fish stocks resulting from factors such as climate change threaten the current regulation on fishery resources allocation (Baudron et al., 2020; Fernandes and Fallon, 2020).

Moreover, the EU recently adopted several initiatives to promote Blue Growth. This project aims to increase the profitability of economic activities performed in marine areas by encouraging new technologies that will ensure the sustainability of the environment. Given their capacity for innovation and generation of new jobs, the sectors primarily considered in this initiative are: ocean energy generation, aquaculture, tourism, biotechnology and exploitation of marine mineral resources (EC, 2017, 2012). The omission of fishing from this plan, particularly the activity of the small-scale fleet, has raised concerns, due to the large reliance on fishery resources and the difficulties in re-allocating a labor force historically dependent on fishing (Said and MacMillan, 2020; Cisneros-Montemayor et al., 2019; Da-Rocha et al., 2019; Boonstra et al., 2018). A lack of references to social aspects -such as equity- has also raised concerns that it may compromise sustainable development (Österblom et al., 2020; Cisneros-Montemayor et al., 2019).

Within the European region, Brexit is a clear recent example of the relevance of equity in the distribution of resources. Unfavorable and unequal distribution of fishery resources in recent years was one of the key arguments for the UK's leaving the European Union (Agnisola et al., 2019; Forse et al., 2019; O'Higgins and O'Hagan, 2019; Napier, 2016). However, this is not the only case of tension associated with unequal distribution between MSs (and former MSs). Garza-Gil and Varela-Lafuente (2015) describes how different potential frameworks for regulating fishing activity under the CFP would affect either small-scale or industrial Spanish fisheries. Each of these fleet segments would prefer to see the regulation that is less harmful to their sector being implemented. Thus, implementation of a single policy favors one fleet segment over the other. Matić-Skoko and Stagličić (2020) highlight the strong opposition to the CFP among small-scale Croatian fisheries, who claim that the regulation does not take into account geographical and socioeconomic idiosyncrasies in fishing areas and communities. They believe CFP benefits small-scale fisheries from other regions of Europe, but not their own. McLean and Gray (2009) compare different perceptions of CFP implementation in Great Britain and Germany. The results show very different expectations of CFP implementation between the two countries. Whereas in Great Britain the regulation is seen as a tool for fulfilling the needs of EU's fisheries, in Germany the CFP is perceived as an instrument for achieving their national goals. Morton et al. (2016) outline the difficulties in allocating fishing activity among different stakeholders in the wild Atlantic salmon fisheries in Scotland, where each stakeholder believed themselves to have a better justification than the others for exploiting these scarce fishery resources.

The present analysis aims to contribute to the reduction of fishing pressure pursued by the CFP while ensuring that fishing remains an active industry within the Blue Growth strategy. For this purpose, it focuses on two aspects. Firstly, it examines whether the heterogeneous distribution of the value of landings per capita between EU countries is motivated by idiosyncrasies in their harvesting areas or by differences in fishing activity. Secondly, it compares the contribution of each production factor to the heterogeneity of the value of landings per capita of countries within the main EU fisheries (the ATW and the MBS areas). Combining these questions, the analysis presents evidence on ways in which fleet capacity reduction could be efficiently proposed in each of the main fishing areas while ensuring homogeneous distribution of fishery resources between countries.

From a methodological viewpoint, the analysis is based on different decompositions of the Theil inequality index with a parameter value of 0 (Shorrocks, 1984, 1980). Inequality metrics have already been implemented to analyze the distribution of multiple natural resources and environmental capacities such as CO2 emissions (Remuzgo and Sarabia, 2015; Duro, 2012; Padilla and Serrano, 2006), coal use (Chen et al., 2018), water use (Hu et al., 2016) and ecological footprint (White, 2007). By contrast, fewer studies exist based on inequality metrics in the field of fishery resources. Among such studies, the research question is very diverse in nature: the distribution of resources in a particular fishery between different fleet segments and its relationship to total production (Baccante, 1995); the concentration of market power among a few large fishers caused by the ITQ system (Abayomi and Yandle, 2012); and the consequences of diverse quota management methods for different fleet segments (Bellanger et al., 2016). Despite applying the same type of methodology, the present analysis focuses on the heterogeneity between countries' fleets rather than between fleet segments of the same fishery. In this regard, this analysis more closely resembles the study in Gutiérrez and Inguanzo (2019), which examines inequality in high seas catches of countries in the period 1960-2014. Despite the decrease in such heterogeneity during the period analyzed, the authors found that at the end of the period, major dissimilarities persisted between countries, mainly due to technological factors. Following this evidence, the present analysis initially decomposes heterogeneity in the per-capita landings of MSs between and within major fishing areas. The two components of heterogeneity give an interesting insight into the reasons for total heterogeneity. On the one hand, heterogeneity between fishing areas reflects dissimilarities caused by the ecological features of each area. On the other hand, heterogeneity within fishing areas results from technological dissimilarities between fleets. After isolating the technological inequality, the multiplicative factor decomposition (Alcalde-Unzu et al., 2009) was analyzed to estimate the contribution of each production factor (technological productivity, technical progress, capital-intensity and fishers per capita) to heterogeneity.

The dataset used for this analysis contains information from 23 MSs for the period between 2008 and 2016 (including the United Kingdom, which was still a Member State during the period analyzed). Several additional aspects should be noted in consideration of this sample. Firstly, the availability of data for certain countries explains their later incorporation to the sample (France has been included from 2010 onwards, Croatia from 2012 onwards and Greece from 2014 onwards). Secondly, the poor quality of data-indicators for some of the countries is an obstacle to comparison (Christou et al., 2019). Moreover, there are several factors that should be mentioned regarding the structure of the analysis and data used. Although the analysis considers the size of countries when weighting the allocation of resources by population, other criteria may underlie their current allocation of fishery resources (Orach et al., 2017; Carpenter et al., 2016). Since the analysis focuses on the distribution of EU resources, it omits the operation of foreign countries in the same fishing areas, which may affect fishing activities in several dimensions such as the status of the stocks (Colloca et al., 2017) and stakeholder participation in management (Coers et al., 2012).

The present analysis shows that more than 90% of the heterogeneity in landings per capita of MSs between 2008 and 2016 is due to heterogeneity in their fishing activity (the within component) rather than to particular features of their fishing areas (the between component). Heterogeneity in ATW represents the largest share of total within inequality (between 50 and 60%). By contrast, heterogeneity in areas other than ATW and MBS (Other Areas) represents less than 1% throughout most of the period analyzed. The negative trend in total heterogeneity reflects the fact that countries' landings homogenized by more than 26 percentage points between 2008 and 2016. This reduction in heterogeneity was mainly caused by the decrease observed in heterogeneity within areas (almost 27 percentage points). The largest decrease in heterogeneity was observed in the MBS area, whose contribution to the within component was reduced by around 38%. In the ATW area, the contribution to the within component decreased by 22%.

Since most of the value of EU landings during the period analyzed (around 84%) comes from the ATW and MBS areas, with Other Areas contributing in less than 1% to within heterogeneity, the multiplicative factor decomposition was only applied to the ATW and MBS areas. This decomposition reflects the fact that the contribution of production factors to heterogeneity within areas varies significantly. The only exception that can be appreciated in both areas is the large contribution of the labor factor. Indeed, fishers-per-capita is the largest contributor to the heterogeneity in both areas. In the ATW area, factors related to technology have a much smaller effect (technical progress) or even contribute negatively to heterogeneity (technological productivity and capital-intensity). In MBS, the second largest contributor to heterogeneity is the technological productivity although its contribution is far behind that of the labor factor. From 2008 to 2016, the contribution of the labor factor has increased noticeably in both areas. By contrast, the greatest decrease was experienced in the contribution of technological productivity.

Based on the evidence of this analysis, the adjustment in fishing capacity sought by the CFP could be advanced by impacting the labor force and KWt per vessel in the ATW area and the labor force and technological productivity in the MBS area. Since lower fishing pressure is considered necessary for the conservation of marine ecosystems, these factors could be homogenized using countries with medium or low landings levels as a benchmark. Special care should be taken with regard to the fishing labor force factor, since the fishing industry is a major support for coastal communities. In absolute terms, the proportion of fishers in the labor force fell in all countries, mainly but not exclusively as a result of aging and a lack of replacement by younger workers. This has even led to the creation of professional fishing schools such as *Enaleia* in Greece and *Instituto de Pesca Marítima del Atlántico* in Spain. Nonetheless, this decrease is more significant, in relative terms, in Northern European countries where young fishers face difficulties entering the industry because of the large capital investment required in acquiring vessels and quotas than in Southern countries (Pita et al., 2020). Considering equity in the burdens of conservation will help to avoid the tensions that have arisen between countries under unequal distributions of resources, increasing the long-term probabilities of the CFP's success. Implementing flexible quota managament (Harte et al., 2019; Hoefnagel et al., 2015) or increasing the involvement of fishers in the regulation process (Williams et al., 2018; Garza-Gil et al., 2017; Sampedro et al., 2016) could help achieve sustainability of the marine environment whilst also keeping the fishing industry alive.

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Appendix 2.A OFR landings

The tables 2.A.1, 2.A.2, 2.A.3, 2.A.4 and 2.A.5 show the percentage of the value of 2008-2016 landings coming from each OFR for Spain, France, Lithuania, Portugal and Italy. For the purposes of clarity, information on OFRs that represent less than 1% of the value of landings during the period analyzed has been omitted.

Table 2.A.1: Origin of OFR landings value 2008-2016, Spain.

Subregion	% OFR Landings (in Euros)	FAO area
51.5	21.00	Somalia, Kenya and Tanzania
41.3.2	11.71	Southern Patagonian
41.3.1	7.05	Northern Patagonian
34.3.1	5.84	Cape Verde Coastal
34.1.3	5.43	Sahara Coastal
77	4.95	PACIFIC, EASTERN CENTRAL
87.2.6	3.07	Central Oceanic
34.3.3	2.58	Sherbro
34.4.2	2.31	Southwest Oceanic
51.6	2.26	Madagascar and Mozambique Channel
34.4.1	2.24	Southwest Gulf of Guinea
34.3.6	2.13	Southern Gulf of Guinea
34.1.2	1.94	Canaries/Madeira Insular
51.4	1.91	Eastern Arabian Sea, Laccadives
47.1.1	1.68	Cape Palmeirinhas Division
41.2.4	1.66	Central Oceanic
47.1.3	1.56	Cunene Division
34.3.4	1.37	Western Gulf of Guinea
47.1.2	1.37	Cape Salinas Division
71	1.35	PACIFIC, WESTERN CENTRAL
81	1.33	PACIFIC, SOUTHWEST
87.1.4	1.30	Northern Oceanic
34.3.2	1.20	Cape Verde Insular
51.7	1.13	Oceanic
51.8	1.12	Mozambique
47.1.5	1.07	Orange River Division

Table 2.A.2: Origin of OFR landings value 2008-2016, France.

Subregion	% OFR Landings (in Euros)	FAO area
51.5	26.28	Somalia, Kenya and Tanzania
34.3.6	20.03	Southern Gulf of Guinea
31	19.61	ATLANTIC, WESTERN-CENTRAL
51.6	14.34	Madagascar and Mozambique Channel
51.7	5.08	Oceanic
34.3.4	3.46	Western Gulf of Guinea
34.3.3	2.28	Sherbro
34.4.1	2.08	Southwest Gulf of Guinea
34.3.1	1.38	Cape Verde Coastal
41.1.1	1.23	Amazon
47.1.2	1.10	Cape Salinas Division
34.4.2	1.06	Southwest Oceanic

Table 2.A.3: Origin of OFR landings value 2008-2016, Lithuania.

Subregion	% OFR Landings (in Euros)	FAO area
34.3.1	22.52	Cape Verde Coastal
34.1.3	18.40	Sahara Coastal
34.1.3.2	13.23	Subdivision 34.1.32
87.2.6	11.43	Central Oceanic
87.3.3	5.85	Southern Oceanic
34.1.1	4.64	Morocco Coastal
34.3.1.1	3.92	Subdivision 34.3.11
34.1.3.1	3.69	Subdivision 34.1.31
27.14.b	2.50	Southeast Greenland
47.1.3	2.32	Cunene Division
27.1.a	2.00	Barents Sea - NEAFC Regulatory Area
27.4.a	1.71	Northern North Sea
27.7.b	1.15	West of Ireland

Table 2.A.4: Origin of OFR landings value 2008-2016, Portugal.

Subregion	% OFR Landings (in Euros)	FAO area
34.1.2	31.20	Canaries/Madeira Insular
51.8	12.34	Mozambique
41.2.4	7.35	Central Oceanic
34.3.1	4.90	Cape Verde Coastal
41.3.1	4.18	Northern Patagonian
51.7	3.98	Oceanic
34.4.2	3.48	Southwest Oceanic
34.1.3	2.42	Sahara Coastal
47.c.1	2.22	47.C.1 SEAFO Division
34.3.3	2.18	Sherbro
41.3.3	2.14	Southern Oceanic
57.3	1.94	Central
34.2	1.94	Northern Oceanic
47.b.1	1.83	47.B.1 SEAFO Division
81	1.81	PACIFIC, SOUTHWEST
34.3.2	1.61	Cape Verde Insular
47.a.1	1.44	47.A.1 SEAFO Division
41.2.3	1.33	Platense
51.6	1.33	Madagascar and Mozambique Channel
41.1.4	1.25	Nothern Oceanic
34.4.1	1.07	Southwest Gulf of Guinea

Table 2.A.5: Origin of OFR landings value 2008-2016, Italy.

Subregion	% OFR Landings(in Euros)	FAO area
34.1.3	53.16	Sahara Coastal
34.3.1	21.59	Cape Verde Coastal
51	15.94	INDIAN OCEAN, WESTERN
34.3.1.3	6.16	Subdivision 34.3.13
34.3.1.1	1.61	Subdivision 34.3.11
34.3.3	1.54	Sherbro

Appendix 2.B Concerning the Between - Within Theil Decomposition

The following are the main steps for decomposition of the inequality in landings per capita for a given fishing area g:

$$\begin{aligned} T(LPC) =& \sum_{i=1}^{N} p_i \ln\left(\frac{\overline{LPC}}{LPC_i}\right) \\ =& \sum_{g=1}^{G} p_g \sum_{i=1}^{n_g} p_{gi} \ln\left(\frac{\overline{LPC}_g}{LPC_{gi}}\right) + \sum_{g=1}^{G} \ln\left(\frac{\overline{LPC}}{\overline{LPC}_g}\right) \\ =& \sum_{g=1}^{G} p_g T(LPC_g) + \sum_{g=1}^{G} p_g \ln\left(\frac{\overline{LPC}}{\overline{LPC}_g}\right) \\ =& T_W + T_B, \end{aligned}$$

where g denotes each of the G groups (fishing areas in this setting), n_g is the number of countries harvesting in the fishing area g, p_g is the proportion of population attributed to gth fishing area, $T(LPC_g)$ is the inequality index calculated taking into account the population of fishing area g and, $\overline{LPC_g}$ refers to the overall average for landings per capita in the gth fishing area.

Appendix 2.C Concerning the Multiplicative Factor Theil Decomposition

Proof of the factor decomposition for the Theil index is provided by Alcalde-Unzu et al. (2009) for a general setting where a variable of interest can be written as the product of m factors without assuming independence, where μ is the overall mean and $\{\mu_{jf}\}_j = 1^m$ are the means of the factors. Instead of deriving the expression corresponding to a given g fishing area, for the shake of simplicity the subindex g has been dropped without loss of generality. We have adapted the decomposition to our application here. Given $LPC_i = LPK_i \cdot KPV_i \cdot VPF_i \cdot FPP_i$, the decomposition is expressed as follows:

$$\begin{aligned} T(LPC) =& T(LPK) + T(KPV) + T(VPF) + T(FPP) \\ &+ \ln\left(\frac{\overline{LPC}}{\overline{LPK} \cdot \overline{KPV} \cdot \overline{VPF} \cdot \overline{FPP}}\right). \end{aligned} \tag{2.5}$$

The Theil index can thus be decomposed into two different components: the sum of each factors' Theil index and an additional term, usually referred to as an interaction term which measures the combined dependence of all factors. Note that factors interact with each other by construction, so $\overline{LPC} \neq \overline{LPK} \cdot \overline{KPV} \cdot \overline{VPF} \cdot \overline{FPP}$ and then T(LPC) $\neq$ T(LPK) + T(KPV) + T(VPF) + T(FPP).

Based on Equation (2.5), it can be observed that each T(F) (F=LPK, KPV, VPF, FPP) measures the partial contribution of factor F to overall inequality while the rest of the factors remain unchanged. The total contribution of factor F to overall inequality is obtained by adding the common part to the partial contribution, T(F) + ln $\left(\overline{LPC}/\overline{LPK} \cdot \overline{KPV} \cdot \overline{VPF} \cdot \overline{FPP}\right)$. These contributions are not necessarily non-negative unless factors are independent (see Shorrocks (1982)), which is not the case here. Obviously the sum of all total contributions exceeds the overall inequality because the common part is taken into account as many times as there are factors. Given the nonlinearity of the dependence term it is not possible to decompose it into additive functions of the factors. Thus, one way of presenting results is to show the partial contribution plus the common contribution due to the interaction term. Another way, used in many inequality decompositions where there is a common part is to attribute the same proportion of the common part to each factor (Jusot et al., 2013; Shorrocks, 1982).

Although (2.5) is the most compact expression for the decomposition, there has been a considerable interest in looking within the dependence component to obtain the factors underlying it. Here we follow the method used in Cheng and Li (2006) and Alcalde-Unzu et al. (2009) to show the main role of the relationships between the factors within this term.

For this purpose, we focus on the overall mean, $\overline{LPC}$, which can be expressed as the sum of the covariances between the production factors. In particular,

$$\begin{aligned}\overline{LPC} &= \sum_i p_i LPC_i = \sum_i p_i LPK_i \cdot KPV_i \cdot VPF_i \cdot FPP_i \\ &= \sum_i p_i LPK_i \cdot KPV_i \cdot VPF_i \cdot FPP_i \pm \overline{LPK} \cdot \sum_i p_i \cdot KPV_i \cdot VPF_i \cdot FPP_i \\ &= \sigma_{LPK,KPV\cdot VPF\cdot FPP} + \overline{LPK} \cdot \sum_i p_i \cdot KPV_i \cdot VPF_i \cdot FPP_i = \dots \\ &= \sigma_{LPK,KPV\cdot VPF\cdot FPP} + \overline{LPK} \cdot \sigma_{KPV,VPF\cdot FPP} + \overline{LPK} \cdot \overline{KPV} \cdot \sigma_{VPF,FPP} + \overline{LPK} \cdot \overline{KPV} \cdot \overline{VPF} \cdot \overline{FPP},\end{aligned}$$

where $\sigma_{A,B}$ is the weighted covariance between A and B.

By introducing this expression for $\overline{LPC}$ into the last term in Equation (2.5), it can immediately be seen that this term represents the interactions between the production factors:

$$\ln\left(\frac{\sigma_{LPK,KPV\cdot VPF\cdot FPP} + \overline{LPK} \cdot \sigma_{KPV,VPF\cdot FPP} + \overline{LPK} \cdot \overline{KPV}\sigma_{VPF,FPP}}{\overline{LPK} \cdot \overline{KPV} \cdot \overline{VPF} \cdot \overline{FPP}} + 1\right).$$

Finally, if the variable of interest is not the overall inequality (LPC) but the inequality of a given fishing area (LPC$_g$), just replace the variables according to the area of interest.

Chapter 3

Who is taking the catches in Norwegian fisheries? Drivers of fishing activity concentration in Norway[1]

Norwegian fisheries have been exposed to an excessive fishing pressure in the last decades, collapsing in some occasions for this reason. Consequently, different conservation policies have been gradually implemented during this period. Among these policies were the fishing quotas, which limit the access and catches of vessels within each fishery. Although quotas were initially nontransferable, their commerce was gradually allowed to incentive the reduction of the fleet size with certain restrictions to preserve the competitive market structure and safeguard the economic support of the communities depending on these resources. Over time, quota trade restrictions have been lessened, rising concerns on the effective limit to quota concentration. An excessive concentration of fishing quotas may result in devastating socioeconomic consequences for the fish consumers and the fishing communities. The present analysis aims to measure the catches concentration in the Norwegian fisheries from 2001 to 2019 by calculating the Normalized Herfindahl-Hirschman Index. Quota trade restrictions are considered by analyzing jointly the owners for which trade is allowed. Additionally, the reasons for the evolution of concentration within these groups have been analyzed. Results from the analysis reflect an increase in the concentration of catches within most fishing groups during the period analyzed, differing the determinants across them. These findings suggest considering alternative conservation policies that minimize the exit of more fishers, distribute resources evenly and homogenize the fleet of fishing groups to prevent an excessive concentration.

3.1 Introduction

The Norwegian fishing sector has undergone multiple regulations since the 1950s aiming to improve its profitability, ensure the sustainable exploitation of fisheries and maintain the employment generated as some coastal communities had a high dependence on this activity (Årland and Bjørndal, 2002). The growth in productivity experienced within the Norwegian economy as a whole was not initially accompanied by the fishing sector, motivating a transfer of employment to more productive activities. The sharp decrease in the number of fishers rose concerns on the stability of the fishing industry, leading to the implementation of subsidies to incentive the development of the fishing activity as well as its productivity in the 1950s (Hannesson, 2013). The subsidization facilitated the expansion of the fishing activity in less harvested areas and its later intensification

[1] This analysis was partially elaborated during the stay in SINTEF Ocean (Trondheim, Norway).

through technical improvements (Standal and Asche, 2018), resulting in an excessive industry size that threatened the sustainability of fisheries in several occasions (Gullestad, 2021). The collapse suffered by the Norwegian spring-spawning herring stock in the late 1960s was a turning point in the management of Norwegian fisheries, highlighting the need to regulate the fishing pressure exerted by this industry in certain fisheries to secure their sustainability. Consequently, the fishing activity of the purse seine fleet became regulated with overall quotas in the 1970s, which were allocated across particular vessels according to their size. The decrease observed in the Arctic cod in the 1980s evidenced the need to extend the fishing activity regulation to the remaining fisheries (Gullestad, 2021; Hannesson, 2013). As a measure to prevent the collapse of fisheries, Real Time Closures (RTC) were introduced in 1984. This policy enforces the temporal closure of a particular fishery when there is a significant amount of bycatches or catches below the minimum size to allow for the recovery of the exploited stock. The recovery of the stocks achieved with the RTC was threatened again some years later by the behavior of fishers, who returned to the waters the catches that reported them lower economic value. Aiming to reach a long-lasting recovery of fish stocks, a discarding ban was implemented in 1987 (Gullestad et al., 2015). Further policies targeting the conservation of fishery resources through the limitation of the industry size were implemented in the 1990s and 2000s, when fishing quotas were gradually introduced in the remaining fleet segments (Gullestad, 2021; Hannesson, 2013).

Overall catch limits for each fishery and species (Total Allowable Catches, TACs) are established annually according to the assessment of independent marine research on the conditions of the fish stocks. TACs are distributed among major fishing groups according to multiple criteria as the employment generated, their settlement and the use of resources. Quotas are further distributed across vessels following the criteria established within the fishing group. On the one hand, group quotas may be distributed across vessels considering technical features like the size and capacity of the vessels, these catch limits for the vessels are known as Individual Vessel Quotas (IVQs) (Norwegian Fisheries Directorate, 2015). On the other hand, group quotas may be distributed across vessels indicating a maximum level of catches for each of them. This practice distributes across vessels a larger amount of catches than the corresponding group quota, ending the fishing activity when the catches of vessels reach the group quota. As a result, maximum catches are not guaranteed for vessels. On the contrary, it creates a competitive environment across fishers that may not coincide with the economic and biological optima (Hannesson, 2013; Årland and Bjørndal, 2002). Even though quotas were not intended to be traded, since 1996 their commerce has been gradually allowed in the different fishing groups. The regulation of these already occurring transactions seemed convenient to facilitate the reduction of the fleet overcapacity while preventing an excessive concentration of the fishing industry. For this purpose, trade restrictions are established regarding the fleet segment, the region, the amount of quota held and the longevity of the transacted quota (Hannesson, 2013). The increase in the limit of quotas held and their longevity regulated over time has allowed a further reduction of the fleet capacity as well as a larger concentration of the fishing activity (Standal and Asche, 2018).

Decreasing the fishing industry size may contribute positively to the preservation of fish stocks (Gullestad, 2021), but may have significant negative consequences in the social and economic fields as well (Årland and Bjørndal, 2002). Indeed, the Norwegian fishing industry obtained catches for a value of 21.6 billion NOK in 2019 and provided full-time employment to around 8700 fishers (Norwegian Fisheries Directorate, 2019a). Similarly, the concentration of the fishing activity may be desirable from the efficiency perspective (Årland and Bjørndal, 2002), but may imply significant costs at social and economic levels. Concentration in the supply side of the Norwegian fishing industry not only shrinks the consumer welfare, but may transfer to the labor market reducing the job opportunities and lowering wages if dominant firms use their market power (Hannah and Kay, 1977). Inequity in the distribution of resources is associated with larger difficulties in the implementation of policies for their conservation (Hori, 2015; Klein et al., 2015) as well as with hindrances to the sustainable development (Österblom et al., 2020; Cisneros-Montemayor et al., 2019). The present analysis aims to measure the economic consequences of the quota concentration experimented in the Norwegian fisheries by looking at the concentration in the value of catches during the last two decades. In order to better adapt to the current regulatory framework, the analysis focuses separately on the evolution of the concentration within each major fishing group. Additionally, the reasons for the evolution of concentration are examined to prevent an excessive concentration.

Concentration has already been analyzed for the Icelandic fisheries (Agnarsson et al., 2016) and the herring and salmon fisheries in British Columbia (Haas et al., 2016). The former studies whether licenses have concentrated from 1993 to 2012 deepening in the nature of owners in which licenses have concentrated during this period. The latter focuses on the concentration of quotas from 1990 to 2014 paying attention to the type of quota holders and their location. Both analyses measure concentration through an array of indexes widely used in the industrial organization (US Department of Justice and Federal Trade Commission, 2010; EC, 2004) and environmental distribution analyses (Inguanzo et al., 2021; Gutiérrez and Inguanzo, 2019; Remuzgo and Sarabia, 2015; Duro, 2012). Along the same lines, this analysis uses the Normalized Herfindahl-Hirschman Index (NHHI) (Brezina et al., 2016; Herfindahl, 1950; Hirschman, 1945) to measure the concentration of the catches value in Norwegian fisheries from 2001 to 2019. Focusing on the value of catches rather than on the fishing quotas or licenses allows to evaluate directly the consequences of the quotas concentration evidenced in Hannesson (2013) and Standal and Asche (2018). The reasons underlying the evolution of the catches concentration within each fishing group are analyzed with econometric analysis (Wood, 2011; Hastie and Tibshirani, 1990).

The structure of the present analysis is as follows. Section 3.2 presents the sources of data for the analysis, its treatment and a broad idea of the catches concentration evolution from 2001 to 2019 at owner and fishing group levels. Section 3.3 describes the procedures used to determine the factors motivating the evolution of the catches concentration within the major fishing groups. Section 3.4 shows the factors determining the catches concentration within each fishing group. Section 3.5 discusses some aspects defining the perspective of the analysis. Finally, Section 3.6 summarizes the context and main findings of the present analysis.

3.2 Description of data

Data for this analysis has been extracted from the Norwegian Directorate of Fisheries, which provides open micro-data on different aspects of Norwegian fisheries such as vessel's ownership (Norwegian Fisheries Directorate, 2020), licenses (Norwegian Fisheries Directorate, 2020) and catches (Norwegian Fisheries Directorate, 2019b). Information of vessels from the different data sets is gathered using the internal vessel identification of the Fisheries Directorate jointly with the ID of the vessel in the register.

The owners data set (Norwegian Fisheries Directorate, 2020) contains information of the owners of vessels in the Norwegian fleet each year specifying also the ownership quota of each one. When the ownership of a vessel is split between multiple owners, owners of the same vessel have been gathered to avoid accounting for each observation of the respective vessel multiple times. The ownership quota has been used to adjust proportionally the catches of owners who share a vessel with incomplete information on all its owners.

The licenses data set (Norwegian Fisheries Directorate, 2020) details the fishing permission of each vessel. There are more than 60 types of licenses defined according to the fishing area of the vessels (coastal, demersal), their length, the species targeted and the harvesting tool.

The catches data set (Norwegian Fisheries Directorate, 2019b) provides information for each vessel on their hauls. Additional variables on the moment of landing catches, their economic value and the technical features of the vessel used for that haul (number of workers, vessel length, engine power) are specified. The economic value of catches has been reconstructed for this analysis using the most recent specification provided by Norwegian Fisheries Directorate (2019b) to measure it, which includes the payment of landings and the post-payments. Hereafter *catches* will refer to the economic value of the catches landed measured in Norwegian krones as previously described unless specified otherwise.

Identifying the observations of the three data sets among them allows to observe initially the catches of each vessel with their corresponding technical (length, engine power and number of workers) and harvesting information (economic value and date). When the owners data set is considered, the observations of vessels corresponding to the same owner can be aggregated. This allows to obtain the total value of catches and the quantity of inputs (average number of workers, average vessel length, average engine power and number of vessels) used by each vessel owner. Sections 3.2.1 and 3.2.2 are based on this data set. Finally, accounting for the information on the fishing licenses makes possible to aggregate the information of owners with the same license. As a result, the average economic value and input factors of vessel owners with each license type can be known. Section 3.4 is based on this data set.

3.2.1 Concentration of catches value across owners

The decrease in the number of fishers occurred in the Norwegian fisheries from 1950 to 2010 contrasts with the increase observed in the value of catches, suggesting that catches have concentrated during this period (Hannesson, 2013).

Aiming to quantify the concentration observed in the fishing industry over time, the present analysis relies on the Herfindahl-Hirschman Index (HHI) (Herfindahl, 1950; Hirschman, 1945). Being advised for the measurement of horizontal concentration by US Department of Justice and Federal Trade Commission (2010) and EC (2004), the HHI has been widely applied in many sectors including fisheries (Agnarsson et al., 2016; Haas et al., 2016; Abayomi and Yandle, 2012; Stewart and Callagher, 2011). In particular, this analysis applies the HHI to measure the concentration of the catches value across vessel owners. Considering the concentration of catches in monetary value allows to measure directly the economic effects of the variations in the market structure. In addition, focusing on the owners of vessels allows to provide a clearer view on the supply side structure of the Norwegian fisheries market. Technically, the HHI used in this analysis adds the proportion of total catches (catches share) of each vessel owner assigning more relevance to larger shares in order to stress the concentration in these cases.

$$HHI = \sum_{i=1}^{n} \left(\frac{catches_i}{\sum_{i=1}^{n} catches_i} \right)^2, \tag{3.1}$$

where $i, = 1, \ldots, n$ refers to each owner. When catches are equally distributed across the n vessel owners, the value of the HHI is $\frac{1}{n}$. Larger values of this index reflect more concentrated distributions of catches. In the case of extreme concentration, when all catches are from one vessel owner, HHI equals 1. Following the indications provided by the EC (2004) regarding the relevance of the concentration, concentration may be divided in three levels (Brezina et al., 2016). Values of the HHI below 0.1 reflect unconcentrated distributions. Between 0.1 and 0.2, the HHI reflects that the distribution is moderately concentrated. In this case, variations of the concentration larger than 0.025 require special attention. Values of the HHI above 0.2 correspond to highly concentrated distributions. Special attention must be paid when variations of concentration larger than 0.015 occur at this level.

As a concentration index, the HHI applied fulfills the following properties (Hannah and Kay, 1977):

- If the concentration curve of a catches distribution lies above the concentration curve of a different catches distribution, it represents a larger concentration in the former catches distribution.
- Sales transfer Principle. If a new vessel owner enters, concentration varies depending on her catches share rather than decreasing automatically because catches are distributed among one owner more.
- There is a catches share ($s_{critical}$) for which entering owners (j) with a share below it ($s_j < s_{critical}$) decrease catches concentration as derived from the previous property.
- Merger increases catches concentration. If the catches share of some leaving owner is transferred to another owner, catches concentration rises.
- Anti-Gibrat effect reduces catches concentration. If owners with different catches share interchange the same proportion of their catches, the catches concentration decreases as the absolute value of catches received by the owner with lower catches share is larger.
- Gibrat effect increases concentration. If owners with different catches shares undergo the same proportional increase of their catches, the catches concentration increases as owners with larger catches shares receive more catches in absolute terms.
- If an owner with catches share close to zero enters or leaves the market, the concentration of catches is not significantly altered.

In order to have an interpretation of the HHI unaltered by the entrance or leakage of owners, the following normalization has been applied to the HHI (NHHI):

$$NHHI = \frac{HHI - \frac{1}{n}}{1 - \frac{1}{n}}. \tag{3.2}$$

This normalization ranges HHI between 0 (representing n owners with equal catches) and 1 (representing 1 owner with all catches), allowing its comparison over periods with different number of owners (Brezina et al., 2016; Cracau and Durán-Lima, 2016). Since changes in this scale may not be properly appreciated, $NHHI$ has been multiplied by 100, expressing it between 0 (n owners with equal catches) and 100 (1 owner with all catches). In order to assess the magnitude of the concentration, the intervals defined for the HHI have been also applied this procedure.

Figure 3.1 illustrates the evolution of the annual catches concentration across owners measured by the NHHI from 2001 to 2019. The colors represent the intervals for which the distribution may be considered unconcentrated (green), moderately concentrated (yellow) and highly concentrated (red). Even though catches distribution seems to maintain within the unconcentrated segment, its concentration has become five times larger during the period analyzed. This increase in the catches concentration coincides with the concentration

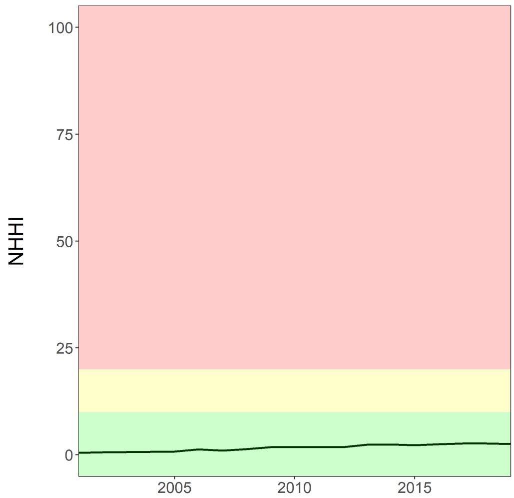


Figure 3.1: Concentration of catches value across vessel owners.

of the fishing activity suggested in Hannesson (2013) and Standal and Asche (2018), which has been associated to an increase of the profitability for vessel owners (Hannesson, 2013) and the sector efficiency (Gullestad, 2021; Standal and Asche, 2018). Nevertheless, a continued evolution of the concentration as the one observed during the period analyzed could rapidly shrinkage the distance to the moderately concentrated level.

3.2.2 Concentration of catches value across owners with similar license types

The depletion of the Norwegian spring-spawning herring stock in the late 1960s constituted a turning point in the Norwegian fisheries management (Gullestad, 2021; Hannesson, 2013). Aiming to recover this stock, the entry to this fleet was limited (Gullestad, 2021). During the 1970s, the activity of the purse seine fleet was regulated in multiple fisheries through catch limits to ensure their sustainability (Hannesson, 2013). Further policies for the sustainable exploitation of Norwegian fisheries were introduced in the 1990s after the deterioration of the Artic cod stock in the late 1980s. Among these policies were the reduction of public subsidies to the Norwegian fishing fleet and the implementation of quotas in the fisheries without them (Gullestad, 2021; Hannesson, 2013).

These quotas are established considering the recommendations of marine research institutions according to the fish stocks conditions. National quotas are allocated across different types of vessels depending on features such as the harvesting area, the fishing tools used, the employment generated and the efficiency in the use of resources. Quotas allocated in groups of similar vessels are then distributed among them according to features such as the vessel size and the tonnage (Norwegian Fisheries Directorate, 2015). Since the 1990s the transferability of quotas across vessels has been gradually introduced for the different vessel types (Standal and Asche, 2018). As quotas are assigned to vessels rather than owners, acquiring them requires from buying the associated vessel. Only when the vessel is withdrawn from the fleet, the corresponding quota can be partially transferred for a limited time to similar vessels (Hannesson, 2013).

These particularities of the quota trade system imply that catches are mainly redistributed across owners with similar vessels. In order to allow for this trade limitation, the present analysis focuses on the concentration of catches occurred into similar groups of vessels.

Table 3.1: Classification of fishing licenses.

Group	License
Coastal	Distriktskvoteordning 2018 (Torsk)
	Driftskvoteordning-Kystflåten
	Kongekrabbe (åpen gruppe), Kongekrabbe (Lukket gruppe)
	Konvensjonelle fartøy < 28 m
	Kystfiskeutvalgskvote - Torsk N62GR
	Kystmakrell-Garn/Snøre < 13m
	Kystmakrell-Garn/Snøre. 13-21.35m
	Kystmakrell-Not < 13 m, Kystmakrell-not. 13-21.35 m
	Leppefisk, Nordsjøsild Notfartøy < 21.35m
	NVG-Kystfartøygruppen, Ringnot 70-90 fot(SUK) Nordsjøsild
	Rognkjeksfiske 13 m og over
	Sei nord.Notfartøy 13 - 27.5m., Sei sør.Notfartøy 13-27.5m.
	Strukturkvoteordning – Kystflåten, Vågehvalfangst
Pelagic trawlers	Kolmuletrålkonsesjon
	Loddetrålkonsesjon, Makrelltråltillatelse
	NVG-trål konsesjon, Pelagisk tråltillatelse
	Slumpfiskordn. Pelagisk Trål, Strukturkv. Kolmuletrål
	Strukturkv.Pelagisk trål, Vassildtrål
Snow crab	Snøkrabbetillatelse
Demersal trawlers	Bonuskvote – Torsketråltillatelse, Distriktskvoter - Torsk N62GR
	Enhetskvote Torsketrålere, Hovedordn.Rederikvote Torsketrål
	Strukturkv. Torsketrål, Slumpfiskordn. Torsketrål
	Torsketråltillatelse, Enhetskv. Grønlandsreketrål
	Grønlandsreketrål, Kystreketrål Sør 11 m og over
	Strukturkv. Grønlandsreketrål, Reketråltillatelse > 65 fot
	Strukturkv. Seitrål, Enhetskvote Seitrålere
	Seitrålkonsesjon, Rødåtetråltillatelse, Flatfiskkonsesjon
Industrial trawlers	Avgrenset Nordsjøtrål, Enhetskvote Industritrål
	Nordsjøtrålkonsesjon, Snurrevad konsesjon
Remote vessels	Fjernfisketillatelse
Conventional	Bonuskvote – Konv.fartøy 28 m og over.
	Brosme og lange-kystfartøy > 28m og < 500m3
	Enhetskv. Konv. Fartøy 28 m og over
	Hovedordn. Rederikvote Konv > 28m
	Konv.fartøy < 28m. Torsk sør for 62gr.
	Konv.fartøy 28 m og over.
	Konv.fartøy >= 28 m Bunnfisk Sør
	Seigarn. Fartøy 28 m og over
	Slumpfiskordn. Konv. fartøy > 28m
	Strukturkv. Konv.fartøy >= 28 m
Purse seiners	Enhetskvote Ringnot > 90 fot
	Hovedordn.Rederikvote Ringnot, Ringnot > 90 fot
	Ringnot 70-90 fot(SUK) Makrell, Seisnurp > 90 fot
	Slumpfiskordn. Ringnot, Strukturkv. Ringnot > 90 fot

Table 3.2: Average monthly composition of vessel groups.

Group	Catches value	Number vessels	Number owners
Coastal	496273326.63	1628.97	1177.93
Conventional	192384575.69	48.44	21.86
Demersal trawlers	776188210.19	195.72	114.94
Industrial trawlers	18466129.45	48.25	47.73
Remote vessels	6353816.26	1.69	1.13
Pelagic trawlers	74833665.39	66.33	39.36
Purse seiners	8033836.10	6.07	3.42
Snow crab	132040354.68	18.08	18.08

Table 3.3: Average inputs used monthly by owners within vessel groups.

Group	Workers	Vessel length	Engine power	Number vessels
Coastal	2.00	12.59	244.57	1.38
Conventional	13.82	42.26	1288.35	2.33
Demersal trawlers	8.76	34.02	2038.82	2.34
Industrial trawlers	4.16	23.72	835.75	1.01
Remote vessels	55.81	100.27	6019.04	1.54
Pelagic trawlers	7.81	36.74	1975.36	1.63
Purse seiners	8.75	48.35	2819.11	1.77
Snow crab	14.35	44.41	2586.71	1.00

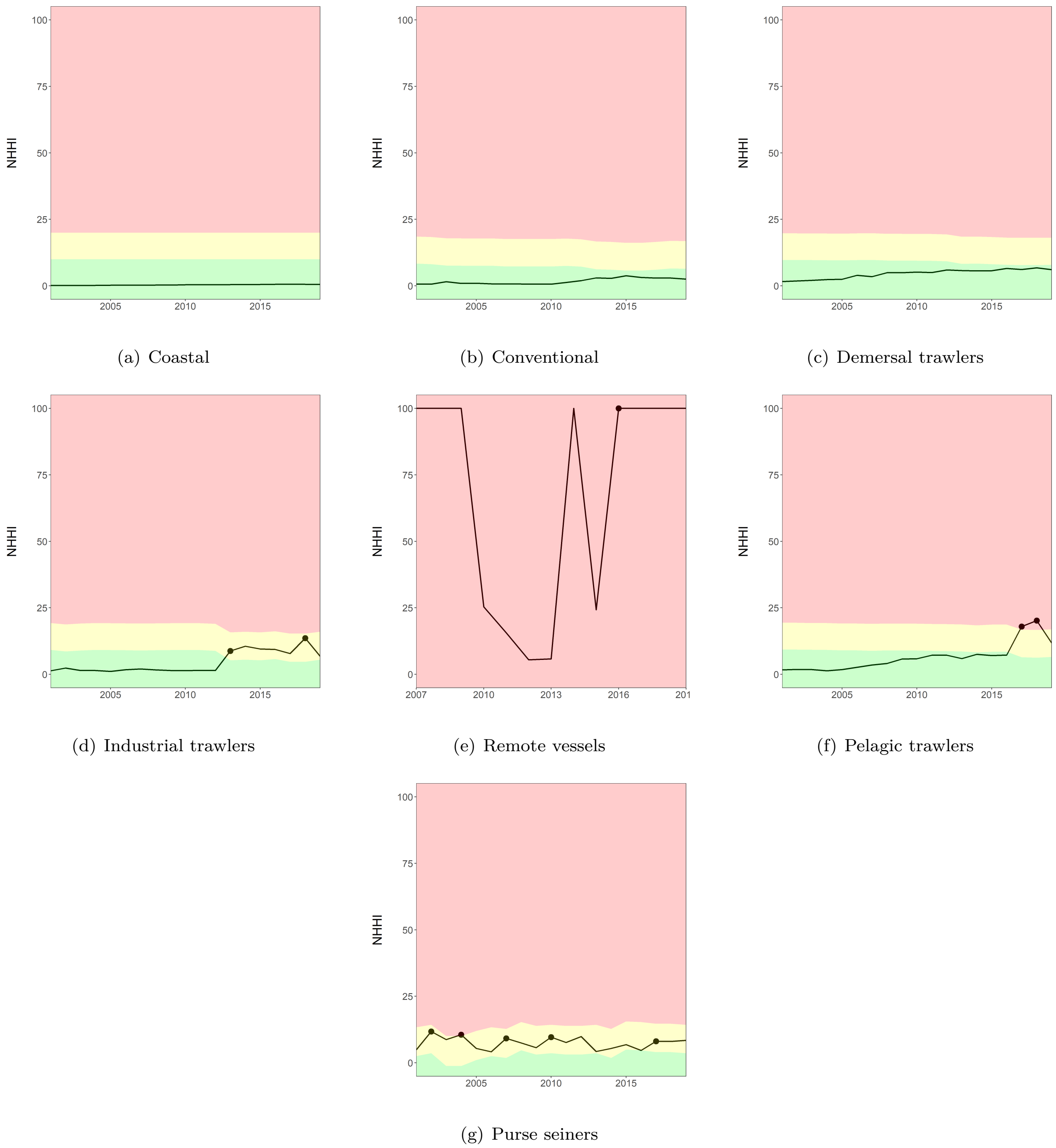
Groups of similar fishing licenses

In particular, fishing licenses have been gathered into groups according to the harvesting area (coastal - most of the catches value during the period analyzed comes from waters within the 12 miles next to the coastline, non-coastal), the fishing gear (trawlers, purse seiners, conventional) and the species caught (demersal, pelagic, crab). Table 3.1 presents the eight groups resulting from combining these criteria. The contribution of vessel groups to average monthly catches is heterogeneous as well as the average number of vessel owners within them (Table 3.2). Almost half of the average monthly catches (45%) comes from *demersal trawlers*, which represent 8% of the vessel owners on average. By contrast, *coastal vessels* account for almost one third of the average monthly catches (29%) and over 80% of the vessel owners on average. Each vessel group characterizes for the use of different factors in their activity (Table 3.3). *Coastal vessel* owners clearly distinguish for employing the lowest number of workers, using the shortest vessels (less than 13 meters) and the lowest engine power (less than 245 horse power). On the contrary, *remote vessel* owners require from the largest number of workers, vessels (over 100 meters) and engine power (over 6000 horse power). For most of the groups, the average owner has no more than 1 vessel. The only exception are owners within the *conventional* and *demersal trawlers* groups, that own an average of 2 vessels.

Due to the limited availability of *snow crab* data (there is only information for year 2019) this group is disregarded in the analyses of the following Sections.

Concentration of catches within groups of similar fishing licenses

Figure 3.2 compares the evolution of the catches concentration within each vessel group from 2001 to 2019. The concentration of the value of catches within vessel groups is measured with the NHHI described in Section 3.2.1, considering n the total number of owners within the corresponding group. The areas colored show the intervals of the NHHI for which distribution may be considered unconcentrated (green area), moderately concentrated (yellow area) and highly concentrated (red area). The dotted values indicate that the variation in concentration that year overpasses the threshold of the corresponding interval to be considered of special relevance. Catches seems to have concentrated from 2001 to 2019 in almost every vessel group although the level of concentration and its evolution differ across them. The *remote vessels*, which experiences the largest levels and variations of concentration, constitutes a special case as its concentration is highly determined by the scarce number of owners within this group (varying from 1 to 3 in some years). The catches concentration of *purse seiners* remains within moderate levels during the period analyzed showing high levels of concentration with special relevance during the first years. Catches concentration of *industrial* and *pelagic trawlers* may be considered unconcentrated during the first decade of the period analyzed approximately, becoming moderately concentrated in the last years. Indeed, *pelagic trawlers* achieve high levels of concentration with special relevance in 2017 and 2018. The lowest levels of concentration are observed in the cases of *coastal* and *conventional vessels* and *demersal trawlers*. Nevertheless, the continued increase in the catches concentration of *conventional vessels* and *demersal trawlers* has approached it close to moderate levels.



(a) Coastal

(b) Conventional

(c) Demersal trawlers

(d) Industrial trawlers

(e) Remote vessels

(f) Pelagic trawlers

(g) Purse seiners

Figure 3.2: Annual concentration of catches within vessel groups.

3.3 Methodology

Since the catches concentration has increased within most of the vessel groups analyzed from 2001 to 2019, discovering the variables beyond such evolution seems necessary to prevent an excessive concentration in the supply side of the Norwegian wild fish production. Bajo and Salas (2002) explain the concentration in a distribution as a result of its inequality and the number of individuals involved. Following this decomposition, the present analysis studies the influence of the number of owners and multiple variables related to catches heterogeneity (such as the distribution of resources and the use of production factors) on the catches concentration.

In order to measure the effect of these variables, Equation 3.3 is specified and estimated using Generalized Additive Models (GAMs) (Wood, 2011; Hastie and Tibshirani, 1990). GAMs allow to estimate simultaneously the parametric or nonparametric effect that a set of variables have over the variable of interest by using penalized maximum likelihood. In particular, Equation 3.3 relates variables of different nature to the monthly catches concentration observed within each group from 2001 to 2019. Four main categories may be distinguished from these variables: the biological conditions of the harvested stocks, the fishing intensity, the number of vessel owners within the analyzed group and time effects. The GAM model specified in Equation 3.3 is estimated using the *mgcv* package developed by Wood (2017) for R assuming a Gaussian family with identity link function.

$$\begin{aligned} NHHI_{it} = {} & \beta_1 \cdot \log\left(Catches_{it}\right) + \beta_2 \cdot \log\left(Catches_{it}\right)^2 + \beta_3 \cdot \log\left(Workers_{it}\right) + \beta_4 \cdot \log\left(Engine\,power_{it}\right) \\ & + \beta_5 \cdot \log\left(Vessel\,length_{it}\right) + \beta_6 \cdot \log\left(Number\,owners_{it}\right) + s\left(t\right) + u_{it}, \end{aligned} \tag{3.3}$$

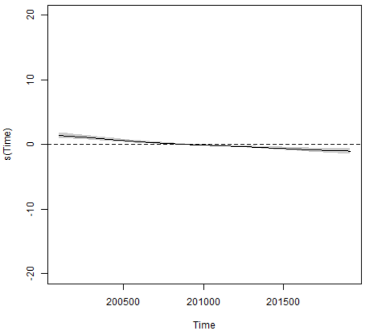
where the subindex it corresponds to the observation of the vessel group $i = (1, \ldots, N)$ in month $t = (1, \ldots, T)$. $NHHI_{it}$ measures the concentration of catches across owners within vessel group i each month t. This index ranges between 0 and 100, taking higher values when the concentration of catches is larger. $Catches_{it}$ refers to the monthly average captures in million krones of the owners within vessel group i. Catches are related to the conditions of the fish stocks, being larger with better conditions (FAO, 2020). Introducing $Catches_{it}$ and its quadratic form into the regression allows to account for the conditions and distribution of the fishery resources within the corresponding vessel group. When β_2 equals zero, β_1 reveals whether the increase in resources is equally distributed (if it is negative) or enlarges inequality across vessel owners (if it is positive). An equal distribution of the resources as indicated in the former case could help to achieve sustainable development whereas the unequal distribution represented in the latter case could harden its accomplishment (FAO, 2020). When β_2 takes values different than 0, the distribution patterns reflected by β_1 are modified with the abundance of the resources. In particular, negative values of β_2 imply that resources are more equally distributed as they become more abundant. By contrast, positive values of β_2 reflect that the distribution of resources becomes more unequal when they become more abundant. When β_1 and β_2 have opposite signs, the effect of the quadratic term offsets the effect of the linear term after a certain level of catches is achieved, showing a total net effect with the same sign as β_2. Given the logarithmic form of the explanatory variables, the level of catches associated to this turning point ($Catches^*$) may be calculated as $Catches^* = e^{\frac{-\beta_1}{2 \cdot \beta_2}}$. $Workers_{it}$ represent the average number of employees by owners within vessel group i at month t. $Engine\,power_{it}$ and $Vessel\,length_{it}$ denote the average horse power and vessel length used by owners within vessel group i at month t. Since variations in the number of $Workers$, $Engine\,power$ and $Vessel\,length$ imply changes in the fishing intensity, their coefficients quantify the impact of differences in the fishing intensity on the expected catches concentration remaining the rest of characteristics equal. $Number\,owners_{it}$ corresponds to the number of owners included within vessel group i each month. Previous literature evidenced a reduction in the number of quota-holders, claiming a consequent increase in the catches concentration (Hannesson, 2013). The coefficient of this variable measures to what extent the decrease in the number of vessel owners observed is responsible for the expected concentration of catches occurred maintaining the value of the rest of variables. By considering these variables in logarithms, their associated coefficients may be interpreted as the percentage change in the expected level of the catches concentration with a variation of 1% in the corresponding variable holding the rest of variables constant. Due to the seasonal nature of the fish stocks harvested as well as the nonlinear annual trend observed in the NHHI evolution for some groups, the time effect has been introduced as a nonparametric term in this model. u_{it} is the error term that accounts for the unobserved factors.

3.4 Results

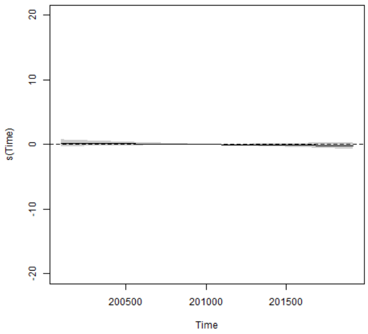
Table 3.4: Non-Parametric estimations.

	Dependent variable: NHHI						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Parametric terms							
log(Catches)	0.359***	−3.937***	3.621***	4.185**	−0.267	1.364	−5.450**
	(0.103)	(0.622)	(0.471)	(1.960)	(1.940)	(1.680)	(2.301)
log(Catches)^2	−0.020	0.813***	−0.445***	0.927	−0.732	−0.506	−0.647
	(0.041)	(0.156)	(0.108)	(0.977)	(0.852)	(0.623)	(0.760)
log(Workers)	−4.144***	−1.810	−16.846***	−9.208	6.695	−10.183**	0.047
	(0.723)	(2.509)	(1.709)	(6.562)	(16.751)	(4.340)	(9.518)
log(Engine power)	4.510***	1.559	−2.588	39.549***	2.031	11.106**	−19.708**
	(1.185)	(1.534)	(1.938)	(8.604)	(8.545)	(4.986)	(8.133)
log(Vessel length)	−3.588	2.447	18.734***	−67.274***	12.564*	−6.991	61.749***
	(2.780)	(3.591)	(4.806)	(18.386)	(7.000)	(11.013)	(19.778)
log(Number owners)	−1.631***	−2.941***	−2.478***	−8.200***	−102.877***	−8.676***	−40.299***
	(0.105)	(0.284)	(0.349)	(1.553)	(5.125)	(1.261)	(2.899)
Non-Parametric terms							
s(t)	4.112***	0.434	0.0004	0.755**	0.002	4.510***	0.776**
Observations	228	228	228	228	117	228	184
Adjusted R^2	0.757	0.534	0.555	0.486	0.844	0.446	0.656
AIC	135.9560	712.267	788.976	1661.287	890.039	1362.277	1708.763
Note:	*p<0.1; **p<0.05; ***p<0.01						
	(1) Coastal, (2) Conventional, (3) Demersal trawlers, (4) Industrial trawlers,						
	(5) Remote vessels, (6) Pelagic trawlers, (7) Purse seiners						

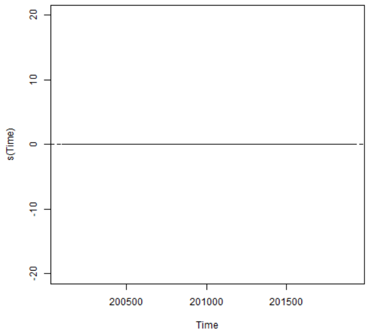
The estimations of model 3.3 are presented in Table 3.4 while the time smoothed term of this model is visualized in Figure 3.3. The results of parametric and non-parametric terms are presented differently. In the



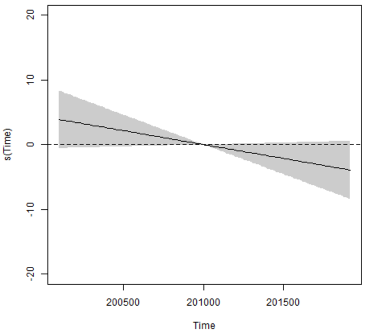
(a) Coastal



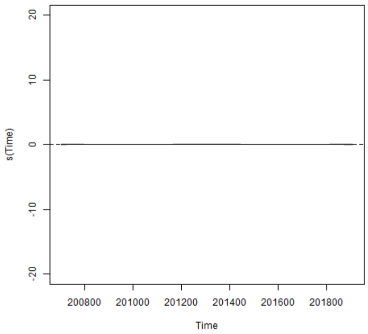
(b) Conventional



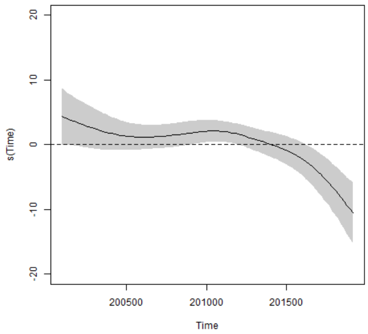
(c) Demersal trawlers



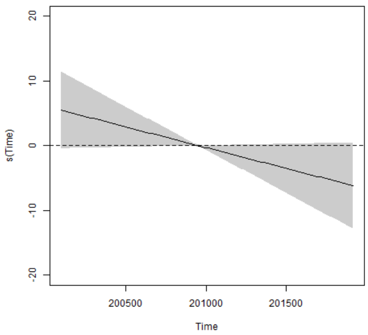
(d) Industrial trawlers



(e) Remote vessels



(f) Pelagic trawlers



(g) Purse seiners

Figure 3.3: Estimated time trend.

former case, the point-wise estimation of the coefficient is shown with its standard deviation in parentheses. In the latter case, the effective degrees of freedom (edf) are indicated. Being related to the smoothness level, the value of the edf reflects the non-linearity of the functional form. In particular, edf close to 0 reflects that the variable is non-significant. When edf equals 1, it reflects that the effect of the variable is linear. Edf between 1 and 2 reflects that there is a weak nonlinear relationship with the variable, which becomes highly non-linear when edf is larger than 2 (Hunsicker et al., 2016; Zuur et al., 2009). The asterisks included after the estimations of both, parametric and non-parametric terms, reflect their significance level.

Estimation results presented in Table 3.4 reveal that the concentration of catches is affected by different variables within each group. In order to proceed with the interpretation of the estimations, it is assumed that factors different from the one specified in each case remain constant.

The estimated coefficients of $\log(Catches)$ and $\log(Catches)^2$ indicate that catches and the index relate differently across groups. In the case of *coastal vessels* and *industrial trawlers*, the relation is linear (since the quadratic term is not significant) and positive, reflecting that resources are unevenly distributed across owners. Thus, improvements in the resources conditions rise catches concentration, which shows that large owners play a major role in the absorption of additional resources. In the case of *demersal trawlers*, both terms are significant. The positive and negative coefficient of the linear and quadratic term, respectively, reflect that improvements in resources conditions contribute to rise catches concentration although this effect gradually decreases, becoming negative when average catches are above 58.81 million NOKs. The distribution pattern changes completely in the case of *purse seiners* and *conventional vessels*. For the *purse seiners*, the relation is linear and negative. This implies that resources are evenly distributed across owners as resources improvements contribute to decrease catches concentration. For the *conventional vessels*, there seems to be a quadratic relation between catches and concentration. In particular, improvements in the resources of this vessel group are associated with reductions in the catches concentration though this effect gradually decreases until average catches are around 11.26 million NOKs, when this effect turns into positive.

The impact of the fishing intensity on the catches concentration within groups depends on the input factor considered. Catches concentration seems to be negatively related to the number of workers in the *coastal vessels* and *demersal* and *pelagic trawlers*. Thus, increasing (decreasing) the average number of workers is associated with lower (larger) levels of concentration in these groups. The effect of engine power on concentration varies across vessel groups. Enlarging (reducing) the average engine power is associated with increases (decreases) in the catches concentration of *coastal vessels* and *industrial* and *pelagic trawlers* and with decreases (increases) in the catches concentration of *purse seiners*. Similarly, the effect of the vessel length differs across groups. Increasing (decreasing) the average vessel length has a positive (negative) effect on the catches concentration of *demersal trawlers*, *remote vessels* and *purse seiners* whereas the effect becomes negative (positive) in the case of the *industrial trawlers*.

The number of owners seems to affect inversely the concentration of catches in every group confirming the findings in previous literature (Hannesson, 2013; Bajo and Salas, 2002). In general, more (fewer) owners are associated to a more even (uneven) distribution of catches.

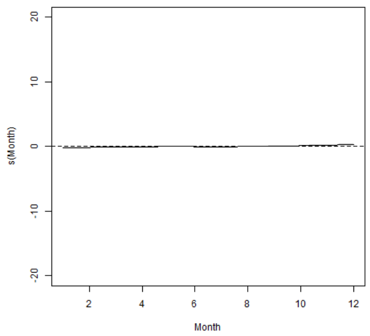
Finally, the significance of the term s(t) indicates that after controlling the effect of the other factors over the concentration index, there is still a significant behavior that varies over time in the *coastal vessels*, *industrial* and *pelagic trawlers* and *purse seiners*. As can be observed in Figure 3.3, this effect is different for each group.

Aiming to identify more precisely the time effect, it is decomposed to account for seasonal and trend effects.

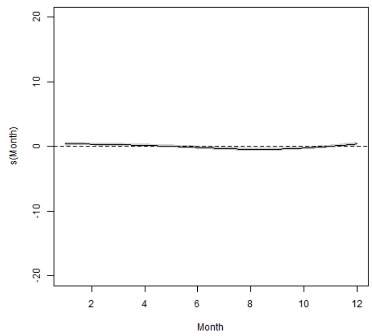
Table 3.5: Non-Parametric estimations with seasonality.

	Dependent variable: NHHI						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Parametric terms							
log(Catches)	0.431***	$-$3.738***	3.145***	6.674***	$-$0.267	3.560**	$-$4.819**
	(0.106)	(0.595)	(0.529)	(2.012)	(1.940)	(1.669)	(2.314)
log(Catches)^2	$-$0.101**	0.752***	$-$0.532***	$-$0.224	$-$0.732	$-$0.572	$-$0.665
	(0.041)	(0.149)	(0.098)	(1.016)	(0.852)	(0.588)	(0.752)
log(Workers)	$-$4.824***	$-$4.568*	$-$15.098***	$-$13.299**	6.695	$-$10.951***	$-$1.932
	(0.768)	(2.546)	(1.775)	(6.524)	(16.751)	(4.189)	(9.509)
log(Engine power)	1.858	0.026	$-$3.912**	40.267***	2.032	10.672**	$-$19.682**
	(1.273)	(1.400)	(1.782)	(8.511)	(8.545)	(4.744)	(8.052)
log(Vessel length)	2.911	7.604**	20.637***	$-$66.518***	12.562*	$-$9.195	62.692***
	(2.942)	(3.358)	(4.470)	(18.138)	(6.999)	(10.652)	(19.559)
log(Number owners)	$-$1.803***	$-$3.352***	$-$2.208***	$-$7.798***	$-$102.879***	$-$5.477***	$-$40.200***
	(0.123)	(0.303)	(0.360)	(1.554)	(5.125)	(1.294)	(2.986)
Non-Parametric terms							
s(Month)	2.936***	2.479***	5.224***	2.161***	0.0003	3.591***	1.171**
s(Year)	7.603***	0.0005	0.819**	0.897***	0.0006	3.553***	0.811**
Observations	228	228	228	228	117	228	184
Adjusted R^2	0.792	0.563	0.665	0.518	0.844	0.493	0.665
AIC	108.708	700.765	732.562	1649.791	890.036	1346.557	1706.292

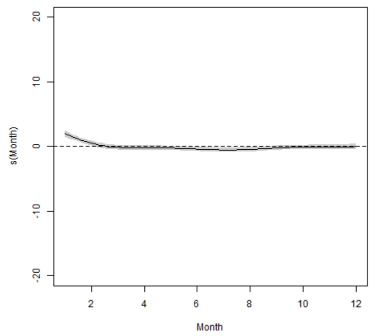
Note: *p<0.1; **p<0.05; ***p<0.01
(1) Coastal, (2) Conventional, (3) Demersal trawlers, (4) Industrial trawlers,
(5) Remote vessels, (6) Pelagic trawlers, (7) Purse seiners



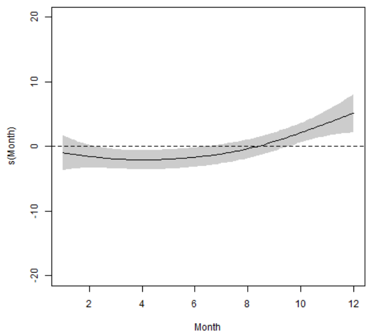
(a) Coastal



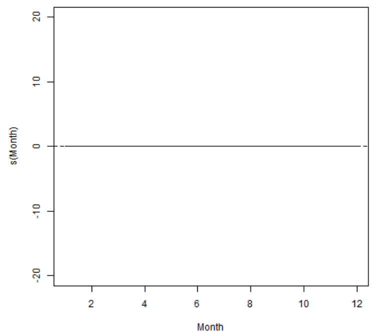
(b) Conventional



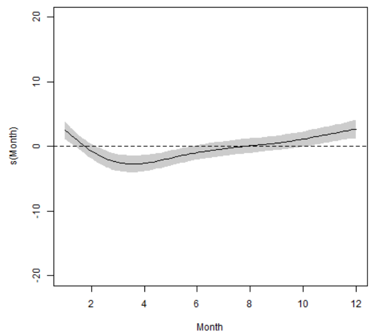
(c) Demersal trawlers



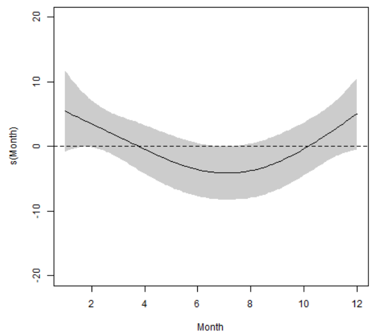
(d) Industrial trawlers



(e) Remote vessels

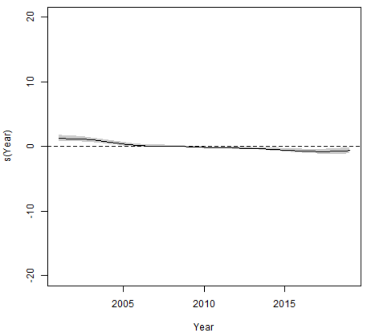


(f) Pelagic trawlers

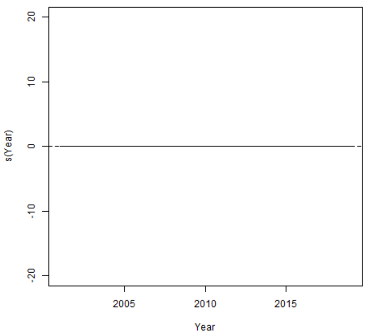


(g) Purse seiners

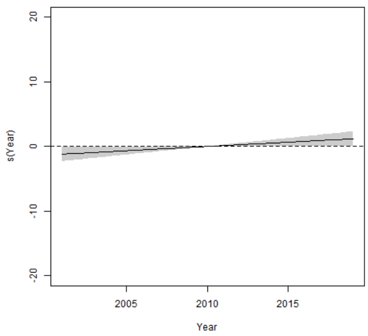
Figure 3.4: Estimated seasonal effects.



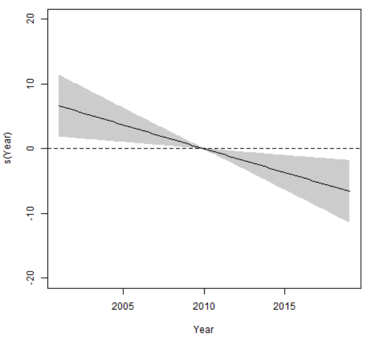
(a) Coastal



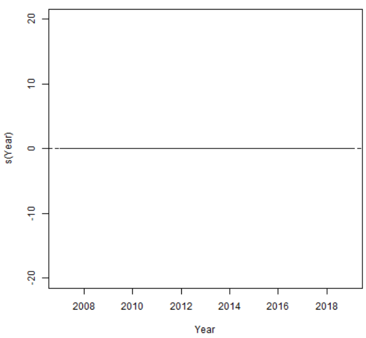
(b) Conventional



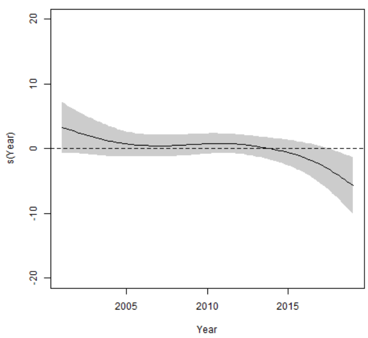
(c) Demersal trawlers



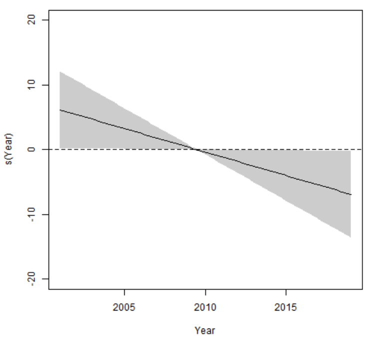
(d) Industrial trawlers



(e) Remote vessels



(f) Pelagic trawlers



(g) Purse seiners

Figure 3.5: Estimated annual trend.

Table 3.6: Parametric estimations.

	Dependent variable: NHHI						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
log(Catches)	0.508***	−3.549***	2.114***	8.362***	0.103	4.095**	−4.462*
	(0.136)	(0.606)	(0.518)	(2.117)	(2.146)	(1.624)	(2.568)
log(Catches)^2	0.004	0.717***	−0.314***	−1.188	−1.146	0.370	−0.836
	(0.041)	(0.161)	(0.096)	(1.095)	(0.981)	(0.569)	(0.829)
log(Workers)	−0.003	−4.008	−5.944***	−6.442	−11.416	−13.231***	−6.868
	(0.716)	(2.925)	(2.069)	(8.000)	(26.092)	(3.838)	(10.667)
log(Engine power)	3.004**	1.812	−7.414***	34.641***	−3.012	−1.999	−17.644**
	(1.342)	(1.917)	(1.723)	(10.423)	(18.424)	(4.153)	(8.915)
log(Vessel length)	−7.777***	−6.197	8.200*	−77.036***	4.379	−5.764	49.408*
	(2.960)	(7.382)	(4.494)	(19.062)	(15.010)	(8.511)	(29.366)
log(Number owners)	−1.796***	−3.355***	−4.312***	−8.906***	−101.386***	−7.957***	−40.986***
	(0.245)	(0.330)	(0.501)	(1.638)	(6.174)	(1.323)	(4.055)
February	0.119	−0.227	−2.062***	−3.130	8.053	−3.021**	−4.107
	(0.126)	(0.377)	(0.389)	(2.971)	(4.920)	(1.484)	(8.910)
March	0.035	−0.366	−1.974***	−5.463*	2.833	−3.681**	−0.512
	(0.157)	(0.366)	(0.400)	(3.019)	(4.894)	(1.665)	(9.508)
April	0.341***	−0.442	−1.645***	−3.469	2.333	−1.750	3.451
	(0.126)	(0.368)	(0.396)	(2.901)	(5.148)	(1.701)	(10.208)
May	0.330***	−0.099	−1.693***	−2.346	5.774	−1.961	−12.783
	(0.112)	(0.375)	(0.401)	(2.915)	(4.989)	(1.514)	(9.140)
June	0.290**	−0.825**	−1.866***	0.959	6.043	−2.583*	−15.156
	(0.114)	(0.381)	(0.394)	(2.906)	(4.878)	(1.463)	(9.232)
July	0.388**	−0.463	−1.654***	−2.916	5.470	−2.741*	−6.889
	(0.167)	(0.369)	(0.386)	(2.992)	(4.825)	(1.455)	(9.214)
August	0.181	−1.022***	−1.825***	−3.186	5.266	−0.751	−13.921
	(0.117)	(0.367)	(0.396)	(2.969)	(4.791)	(1.482)	(8.956)
September	0.430***	−0.931**	−1.549***	−0.327	6.874	−0.667	−11.813
	(0.117)	(0.378)	(0.383)	(2.890)	(4.969)	(1.437)	(9.295)
October	0.464***	−0.970***	−1.783***	2.875	−5.698	−1.398	0.119
	(0.117)	(0.368)	(0.363)	(2.846)	(7.244)	(1.421)	(11.788)
November	0.415***	−0.187	−1.817***	4.651	5.729	1.173	−1.130
	(0.117)	(0.393)	(0.380)	(2.857)	(8.865)	(1.418)	(10.373)
December	0.505***	−0.170	−1.587***	5.491*	1.644	1.076	4.693
	(0.134)	(0.395)	(0.405)	(2.880)	(6.759)	(1.448)	(10.743)
Year	−0.113***	0.048	0.108	−1.100***	0.449	0.067	−0.689
	(0.030)	(0.056)	(0.065)	(0.299)	(0.580)	(0.198)	(0.499)
Constant	244.509***	−58.017	−154.866	2,278.733***	−753.791	−40.964	1,437.425
	(61.156)	(106.107)	(133.410)	(605.671)	(1,086.876)	(392.541)	(963.119)
Observations	228	228	228	228	117	228	184
Adjusted R^2	0.774	0.568	0.722	0.521	0.838	0.541	0.655
AIC	125.504	706.516	694.019	1656.248	906.389	1325.145	1720.276

Note: *p<0.1; **p<0.05; ***p<0.01
(1) Coastal, (2) Conventional, (3) Demersal trawlers, (4) Industrial trawlers, (5) Remote vessels, (6) Pelagic trawlers, (7) Purse seiners

For this purpose, a nonparametric time effect is included for the months and years separately. Table 3.5, Figure 3.4 and Figure 3.5 present the estimation results for this nonparametric model for each vessel group. Additionally, in order to test the adequacy of the non-linearity in the time effect, a parametric specification is estimated and compared to the previous nonparametric estimations to ensure the optimal specification for each vessel group. This parametric model includes dummies for each month (January as reference group) and an annual trend variable (Year). Table 3.6 shows the estimation results for the parametric model. The three specifications are compared for each vessel group using the Akaike's Information Criteria (AIC).

In particular, the nonparametric specification with seasonality seems more appropriate for almost every vessel group with the exception of *demersal* and *pelagic trawlers*. This specification implies some changes in the effect of explanatory variables with respect to the nonparametric specification. With respect to $\log(Catches)$ and $\log(Catches)^2$, their estimated coefficients reflect a different distribution of resources within each of the groups. In the case of *industrial trawlers*, the distribution of resources seems unequal since an increase in fishery resources of 1% leads to an increase in catches concentration of 6.6%. For *coastal vessels*, the distribution of resources seems to be unequal until average catches are around 8.44 million NOKs, when the effect reverses and improvements of resources are associated with a lower catches concentration. *Purse seiners* show a more homogeneous distribution as an increase in their resources of 1% is associated with a decrease in catches concentration larger than 4%. *Conventional vessels* seem to distribute resources evenly until average catches reach the 12 million NOKs, when improvements in their resources are associated with positive variations in the catches concentration. The average number of workers seems to be negatively associated with the catches concentration in *coastal* and *conventional vessels* and *industrial trawlers*. Particularly, increasing (decreasing) the average number of workers in 1% is associated with a decrease (an increase) in catches concentration of more than 4% for the first two groups and more than 13% for the *industrial trawlers*. The engine power seems to be positively related to the catches concentration of the *industrial trawlers* while negatively related to the catches concentration of the *purse seiners*. Thus, a positive (negative) variation of 1% in the average engine power leads to an increase (a decrease) in catches the concentration of *industrial trawlers* larger than 40% and to a decrease (an increase) in the catches concentration of *purse seiners* of 19%. Similarly, the effect of vessel length on the catches concentration varies across vessel groups. Increasing (decreasing) the average use of this factor in 1% produces an increase (a decrease) in catches concentration larger than 7, 12 and 62% for the *conventional* and *remote vessels* and *purse seiners*, respectively. By contrast, if the average vessel length of *industrial trawlers* is increased (decreased) in 1%, its catches concentration decreases (increases) in 66%. As in the nonparametric specification, the number of owners is indirectly related to the level of catches concentration for every vessel group. The nonparametric time terms reflect that the catches concentration of *coastal vessels*, *industrial trawlers* and *purse seiners* is influenced by annual and seasonal factors. However, catches concentration seems only affected by seasonal factors in the case of *conventional vessels*.

The parametric specification seems to be more adequate for the *demersal* and *pelagic trawlers*. Several changes are observed in the effect of the explanatory variables using this specification. The estimated coefficients for $\log(Catches)$ and $\log(Catches)^2$ reflect that catches concentration is linearly related to the abundance of resources for the *pelagic trawlers* while it exhibits a quadratic relationship in the case of *demersal trawlers*. In particular, the distribution of resources in the former group is uneven as an increase in 1% of resources is associated with an increase of 4% in the catches concentration. The distribution of resources in the *demersal trawlers* seems uneven until average catches reach approximately the 29 million NOKs, when resources increases are associated to lower levels of catches concentration. The average number of workers affect negatively the catches concentration of both groups. Thus, increasing (decreasing) the average use of this factor by 1%, catches concentration decreases (increases) in around 5 and 13% in the *demersal* and *pelagic trawlers*, respectively. An increase (a decrease) of 1% in the average engine power used by *demersal trawlers* reduces (enlarges) its catches concentration by around 7%. By contrast, increasing (decreasing) 1% the average vessel length of *demersal trawlers* rises (diminishes) its catches concentration in more than 8%. As in the nonparametric specification, the number of owners affect negatively to the catches concentration of both vessel groups. The month variables reflect that catches become more concentrated in January for the *demersal trawlers* and less concentrated in February, March, June and July for *pelagic trawlers*. Nevertheless, the annual trend seems non-significant for any of these groups.

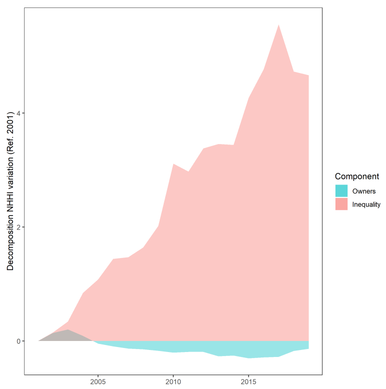
3.5 Discussion

This Section overviews several concerns regarding the selection of the concentration index chosen in the present analysis, the classification of fishing licenses into the vessel groups and the contributions of the present analysis to previous literature.

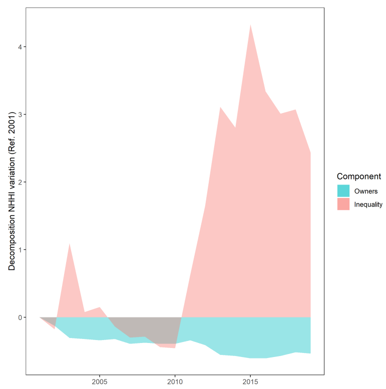
3.5.1 On use of the NHHI

When the concentration of catches is measured with the unmodified HHI, changes in its evolution can be explained by variations in the distribution of catches across vessel owners (inequality component) and variations in the number of owners (sample size component) (Bajo and Salas, 2002). Changes in the inequality component are directly related to changes in the concentration of catches, as an increase (decrease) of the inequality in the distribution of catches implies that catches are more (less) concentrated in certain owners. On the contrary, the sample size component is inversely related to catches concentration. A larger (smaller) number of owners widens (shrinks) the distribution of catches, reducing (increasing) the possibilities to be concentrated in fewer owners. Figure 3.6 illustrates the contribution of each component to the variations in the concentration of catches within groups with respect to 2001 using the decomposition in Bajo and Salas (2002). Given that changes in the number of owners are inversely related to changes in the catches concentration, the sign of the sample size component must be inverted to properly interpret its contribution to catches concentration. Thus, negative (positive) values of the sample size component reflect that the number of owners decreases (increases) and, consequently, the concentration of catches experiences an enlargement (reduction). Both components have contributed positively to the evolution of the catches concentration in most of the vessel groups during most of the years analyzed, especially the inequality component. The only exemption are *purse seiners* and *remote vessels*, where the increase in the number of owners lowered catches concentration, surpassing the positive contribution of the inequality component in the latter group. The NHHI allows to compare concentration levels over time as concentration is adjusted to account for the sample size component.

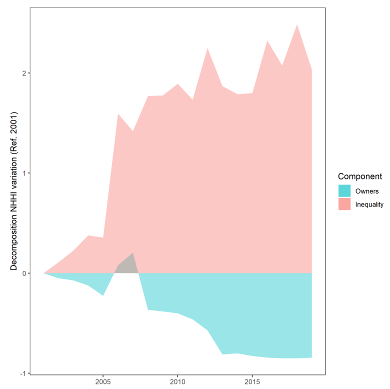
The present analysis relies on the HHI as indicated by the US Department of Justice and Federal Trade Commission (2010) and the EC (2004). Nevertheless, different preferences for stressing the concentration of catches on large owners would imply the use of alternative indexes (Hannah and Kay, 1977).



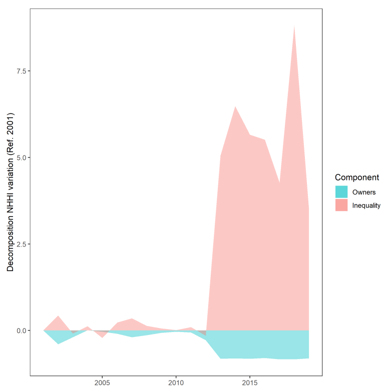
(a) Coastal



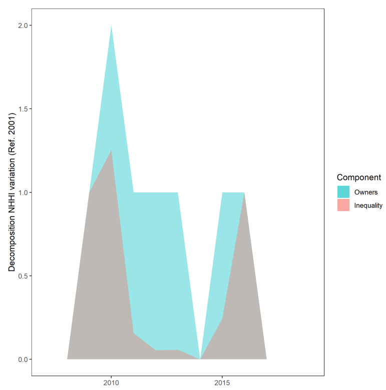
(b) Conventional



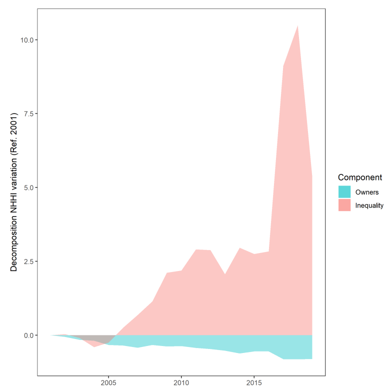
(c) Demersal trawlers



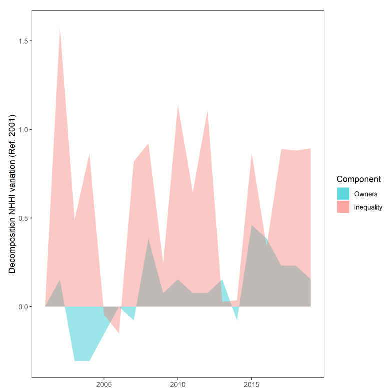
(d) Industrial trawlers



(e) Remote vessels



(f) Pelagic trawlers



(g) Purse seiners

Figure 3.6: Decomposition of the catches concentration within vessel groups.

3.5.2 On the classification of fishing licenses

Gathering the owners with similar fishing licenses allows to provide a more robust and clear perspective on the evolution of catches concentration in the Norwegian fisheries by considering the largest number of observations available while accounting for the quota trade restrictions among them (Norwegian Fisheries Directorate, 2015). In particular, the broad classification of fishing licenses applied in the present analysis aims to emphasize that quotas cannot be traded among extremely different vessels rather than to claim that quotas can be traded among vessels in the same group. In this respect, joining owners with similar vessels into the same group provides a more robust view on the general trend of catches concentration in the Norwegian fisheries. The classification of fishing licenses into similar groups from the present analysis aims to preserve the classification of vessels from previous literature (Standal and Asche, 2018; Hannesson, 2013) in order to facilitate the comparison of findings. Even though certain fisheries are not managed through quotas (Norwegian Fisheries Directorate, 2015), they have been included in this classification to have a more complete view of the Norwegian fisheries. Being considered into the groups with similar characteristics allows to maintain the quota trade restrictions for the quota managed fisheries.

3.5.3 On the context of findings

Equity in the distribution of fishery resources has been recognized as one of the pillars for the sustainable development of fisheries (Österblom et al., 2020). In this respect, the concentration of fishing permits shapes the distribution of catches, producing significant socioeconomic effects. The concentration of fishing permits has already been analyzed using concentration indexes in Agnarsson et al. (2016) and Haas et al. (2016) for the Icelandic and some British Columbia fisheries, respectively. Both studies have found that inequity has increased in the distribution of fishing permits. Norwegian literature has presented evidences on the concentration of fishing quotas in different segment fleets by examining the evolution of the number of vessels with fishing licenses and their quotas (Standal and Asche, 2018; Hannesson, 2013). The present analysis evaluates the concentration in the fishing activity directly on the fishing output in order to assess better its socioeconomic consequences. Rather than looking at the number of vessels, this analysis focuses on the number of owners to provide an overview of the market concentration. Based on the evolution of concentration indexes, it confirms the increase in the Norwegian fishing activity from 2001 to 2019 suggested in previous literature. Indeed, concentration rises during the period analyzed within most of the vessel groups defined to broadly consider quota trade restrictions. The reasons underlying the positive evolution of concentration vary across groups although the larger inequity in the distribution of catches is the main reason for most of the vessel groups. As claimed in Standal and Asche (2018) and Hannesson (2013), changes in the number of vessel owners have contributed significantly to the evolution of catches concentration within groups.

The econometric analysis reflects that catches concentration is also determined by the distribution of resources within groups, the use of fishing inputs and time factors. The uneven distribution of resources contributes to increase the catches concentration of *industrial* and *pelagic trawlers*, *coastal vessels* when average catches are below 8.44 million NOKs, *demersal trawlers* when average catches are below 28.97 million NOKs and *conventional vessels* when average catches are above 12 million NOKs. The average number of workers contributes to decrease the catches concentration of every group except *industrial trawlers* and *purse seiners*. The engine power affects negatively to the catches concentration of *demersal trawlers* and *purse seiners*, but positively to the catches concentration of *industrial trawlers*. The vessel length influences negatively on the catches concentration of *industrial trawlers*, but positively in the catches concentration of the remaining groups except for *coastal vessels* and *pelagic trawlers*. Seasonal effects seem to be relevant in all groups except for *remote vessels* while annual effects are only observed *coastal vessels*, *industrial trawlers* and *purse seiners*.

3.6 Conclusions

Given the difficulties found in the conservation of marine ecosystems under the current exploitation, new international strategies for a more sustainable and efficient use of these resources have been proposed. The Ocean Panel (2021), composed by a group of national representatives including Norway, has designed a plan for the development of existing and new-edge maritime activities in the short term complying with the principles of the UN Sustainable Development Goals for the 2030, previous international environmental agreements and regulatory frameworks. The high socioeconomic impact of the fishing sector in many coastal communities has positioned it among the activities to be promoted in this Strategy. Nevertheless, the environmental damages experienced with the fishing activity patterns in the last decades stress the need of transforming actions to ensure fisheries sustainability. These actions involve the elimination of illegal, unreported and unregulated catches by improving the tracking system of the fishing activity; the facilitation of the cooperation for the efficient distribution of harvested resources; the elimination of fisheries subsidies to reduce the fleet overcapacity and the fish stocks over-exploitation; the minimization of unwanted catches and food waste; the implementation of scientific-based management plans; the empowerment of control organizations and the research for harvesting new species minimizing the environmental effects. Equity in the distribution of resources has been acknowledged as one of the main pillars for the success of these actions (Österblom et al., 2020).

Some of these actions have been targeted by Norwegian fisheries regulatory framework since the 1960s, after the collapse of the Norwegian spring-spawning herring stock. The deterioration of this valuable fishery evidenced the mismatch occurring between the fish stocks conditions and the fleet capacity (Gullestad, 2021). In order to prevent future collapses of this fishery, the purse seine fleet activity was regulated with fishing quotas. After the downsize of the Arctic cod in the 1980s, fishing quotas were gradually introduced in most of the remaining fisheries to ensure their conservation (Hannesson, 2013). Fishing quotas are annually established for each fishery according to the scientific advice on the fish stocks conditions. Total quotas for each fishery are distributed across major fleet segments considering multiple criteria like the employment generated, their settlement and the efficiency in the use of resources. Group quotas are differently assigned across vessels depending on the fishery (Norwegian Fisheries Directorate, 2015). While some groups distribute quotas across vessels following particular criteria such as the vessel length (Norwegian Fisheries Directorate, 2015), others rely on a more competitive framework by only setting the maximum level of catches for each of them (Hannesson, 2013; Årland and Bjørndal, 2002). The latter procedure has been criticized for racing the harvest in the fishery without considering biological nor economic optima as fishers are pressured to take their part before the fishing quota is reached and the fishery closed (Årland and Bjørndal, 2002). Aiming to fasten the reduction in the fleet overcapacity vessel quotas were gradually allowed to be traded in the different fisheries since the 1990s taking advantage of the economic value naturally assigned to these fishing rights. In order to prevent an excessive concentration of the fishing activity that could threaten coastal communities with high dependence on this activity, trade conditions were established. In particular, regional, vessel type, acquisition and durability restrictions were implemented (Hannesson, 2013). The upward modifications of the acquisition limit regulated over time and the lessening of the restrictions have allowed a further concentration of the fishing activity (Standal and Asche, 2018). Indeed, certain vessels seem to have been replaced by larger of their kind in an attempt to the improve efficiency (Hannesson, 2013). The consequences of concentration go beyond the fleet composition (Hannesson, 2013), the market structure and the welfare of consumers (Hannah and Kay, 1977), having wide socioeconomic implications for the communities with a significant dependence on this activity (Österblom et al., 2020; Cisneros-Montemayor et al., 2019).

The present analysis aims to observe the fish market consequences of the quotas concentration evidenced in previous literature by measuring the concentration of the catches value across owners from 2001 to 2019. In this process, the restriction imposed to owners with different group vessels to trade their quotas is broadly considered by analyzing the catches concentration across owners with similar vessels. For this purpose, eight groups of similar vessels have been identified in an attempt to maintain the classification criteria in Standal and Asche (2018) and Hannesson (2013). These groups correspond to: *coastal vessels* (vessels whose largest value of catches during the period analyzed is obtained from the 12 miles close to the coastline), *conventional vessels*, *demersal trawlers*, *industrial trawlers*, *remote vessels*, *pelagic trawlers* and *purse seiners*. Besides quantifying the catches concentration within each group in the last two decades, the analysis deepens in the factors determining its evolution. Each of these targets is achieved with different methodologies. Concentration is usually measured through the Herfindahl Hirschman Index (HHI) (Herfindahl, 1950; Hirschman, 1945), which adds the market shares of the individuals studied emphasizing the concentration in the largest ones. Ranging between the inverse number of individuals (when resources are equally distributed) and 1 (when resources are concentrated in one individual) hardens the comparison of the concentration when the number of individuals varies. Thus, the present analysis relies on the Normalized Herfindahl Hirschman Index (NHHI) as defined in Brezina et al. (2016), which allows to measure the catches concentration accounting for the variations in the number of vessel owners. The factors underlying this concentration are analyzed through econometric regressions.

The positive trend of the NHHI reveals that the value of catches has concentrated in fewer owners from 2001 to 2019, reflecting the quota concentration evidenced in Standal and Asche (2018) and Hannesson (2013). Catches have also become more concentrated within most of the vessel groups defined during the period analyzed. The only exception are *remote vessels*, heavily influenced by the small number of owners sampled. The factors underlying the concentration of catches vary across groups. Nevertheless, the negative impact of the number of owners on the concentration level holds in almost all groups, coinciding with the theoretical effect found in Bajo and Salas (2002). This inverse relationship indicates that the concentration of catches is larger (lower) when there are fewer (more) owners. Catches concentration seems affected by variations in the input factors only in specific cases. Enlarging the average number of workers contributes to decrease catches concentration of almost all groups, being *industrial trawlers* and *purse seiners* the only exceptions. An increase in the average engine power of vessels affects catches concentration positively in the case of *industrial trawlers* and negatively in the case of *demersal trawlers* and *purse seiners*. Increasing the average size of vessels is associated with reductions in the catches concentration within *industrial trawlers* whereas it is associated with an increase in the catches concentration of the rest of groups excepting *coastal vessels* and *pelagic trawlers*. The distribution of resources has a significant effect in every group except in *remote vessels*. In particular, the heterogeneous distribution of resources has a positive effect on the catches concentration of *industrial* and *pelagic trawlers*, *coastal vessels* when average catches are below 8.44 million NOKs, *demersal trawlers* when average catches are below 28.97

million NOKs and *conventional vessels* when average catches are above 12 million NOKs. Monthly and annual variables reveal that catches concentration is affected differently by seasonal and annual effects.

Findings from the present analysis show that the Norwegian fish supply has structurally changed from 2001 to 2019 towards a more concentrated sector, raising concerns on the socioeconomic consequences of this process (Standal and Asche, 2018; Hannesson, 2013). While the experienced concentration has facilitated the reduction of the fleet overcapacity and increased the profitability of the remaining fishers (Gullestad, 2021), it has ended with the job of numerous workers (Standal and Asche, 2018; Hannesson, 2013) and hardened the possibility for young people to enter the sector (Pita et al., 2020). The inherent distribution of resources within the groups defined tends to enlarge the catches concentration as well as the increasing use of fishing inputs in certain cases. If the devastating consequences of an excessive concentration are to be avoided, as initially intended with the quota trade restrictions, further efforts are required to design alternative marine conservation policies that prevent the exit of more fishers, ensure an even distribution of resources and homogenize the fleet of certain groups. These policies are in line with the principles of the Transformations for a Sustainable Ocean Economy signed by Norway (Ocean Panel, 2021), aiming to achieve a sustainable inclusive development of the fishing sector.

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Chapter 4

Reducing waste from the global fishing industry: The case of discards [1]

Discarded catches are a harmful waste from fishing activity that may threaten the equilibrium of the ecosystems and constrain the resources to feed the increasing human population. Although ending hunger and fisheries conservation are recognized as issues needing urgent international collaboration, only a few countries have unilaterally adopted measures to prevent discards. Previous literature has focused on the performance of certain fleet segments in particular fisheries. The present analysis contributes to the design of global policies to minimize discards by showing the socioeconomic and technological factors that determined the discards produced globally from 1961 to 2016. For this purpose, the analysis relies on the estimation of General Additive Models (GAMs). Main findings suggest that discards may be reduced improving the selectivity of certain gear types, re-orienting their activity towards species for which they are more efficient and promoting fish demand.

4.1 Introduction

Discarded catches refer to the organisms of both commercial and non-commercial value that are caught during fishing operations and returned to the waters, often dead or dying (Feekings et al., 2012). These catches are estimated to represent between 9 and 15% of the global catches (Pérez Roda et al., 2019; Gustavsson et al., 2011). According to Pauly and Zeller (2015), global discards were above 8 million tonnes in 2016.

Significant private, social and environmental costs arise from discards. From the private perspective, using inputs to obtain unproductive catches diminishes the profits of fishers (Clucas, 1997), highlighting the need for improvement in the fishing technology. Despite the technological advances in the overall fishing industry during the last decades (Valdemarsen, 2001), large differences in the selectivity of fishing gears persist. In particular, bottom trawlers represent one of the largest producers of discards, discarding over 20% of the catches. By contrast, purse seiners, the largest producers of landings, discard around 4% of the catches (Pérez Roda et al., 2019; Cashion et al., 2018). In this respect, multiple initiatives to improve the selectivity of gears have been proposed (Herrmann et al., 2020; Tang et al., 2019; Vogel et al., 2017). From the social perspective, discards represent an important loss of potential food that could otherwise be used to satisfy the nutritional needs of the increasing human population (FAO, 2018; Diamond and Beukers-Stewart, 2011; Blanco et al., 2007). From the environmental perspective, unwanted catches decrease the size of discarded fish stocks, altering the ecosystem

[1] This analysis has been presented in the *IX AERNA Conference (2021)*, the *35th IWSM (2020)*, the *2020 SAEe*, the *35th EEA Congress (2020)* and the *25th EAERE Annual Conference (2020)*.

balance. In addition, their decomposition alters the conditions of the ecosystem where they are thrown (Clucas, 1997).

Numerous theoretical and empirical studies have already characterized discards in different ways attempting to show some of the multiple factors that should be considered for the minimization of these catches. From the theoretical point of view, most literature has focused on modeling discards as the result of rational decisions based on the cost of landing catches compared to the associated benefits (Pascoe, 1997; Vestergaard, 1996; Arnason, 1994) and the capacity restrictions (Anderson, 1994). Theoretical studies have also analyzed how the framework set by different fishing management regulations allow or incentivize discarding practices (Singh and Weninger, 2015; Hatcher, 2014; Holland, 2010; Abbott and Wilen, 2009; Herrera, 2005; Turner, 1997). From the empirical point of view, previous literature has focused on the main technological and biological factors underlying the discards at micro (vessel) level in specific fisheries (Bellido et al., 2019; Fauconnet et al., 2019; Pulver and Stephen, 2019; Carbonell et al., 2018; Madsen et al., 2018; Maina et al., 2018; Pointin et al., 2018; Maeda et al., 2017; Pennino et al., 2014; Tsagarakis et al., 2014; Feekings et al., 2013, 2012). A small number of empirical analyses has described the main features of global discards like the area of origin or the responsible fishing gears (Cashion et al., 2018; Zeller et al., 2018). Nevertheless, no studies have been performed on the causes of discards at a macro (country) level. By analyzing the determinants of discards at this aggregated level, the present analysis aims to highlight the most prominent issues that should be addressed in order to design effective international policies to decrease discards following the evidences of previous literature.

Even though international cooperation is urgently required for the conservation of fisheries and ensuring food security (UN, 2015), only a few regions have implemented policies preventing discarding practices in their fisheries (i.e. British Columbia, Faroe Island, Iceland, New Zealand, Norway, US Alaska and, more recently, the European Union) (Condie et al., 2014). Market control measures and technical regulations to improve gears selectivity are frequently combined in the discarding regulations implemented (Da-Rocha et al., 2018; Condie et al., 2014). In the case of the EU, region with the largest number of countries committed to the same discarding regulation, the latest reform of the Common Fisheries Policy (EC, 2013) enforce Member States to land all catches, counting them against quotas. However, this type of regulation may seem contradictory as landings of discards are promoted instead of reducing discards at the origin (Sardà et al., 2015). Discarding regulations need to be complemented by further policies ensuring the sustainable exploitation of fisheries and the commitment of stakeholders (Maynou et al., 2018; Condie et al., 2014; Johnsen and Eliasen, 2011). In this regard, the present analysis accounts for the diverse nature of discards by covering not only the fishing technology and socioeconomic variables inherent to the fishing activity, but recognizing also the role played by external factors out of reach from fishers, such as climate conditions. The effect of fishing technology is evidenced comparing the use of different gear types. Social variables involve national and foreign fish demand. Economic variables account for the alternatives in fish production, the ownership of the resources exploited, the industrialization of the fishing entity and the value assigned to catches. External variables like the climate conditions are represented in the time trend. Consequently, discarding policies can be fitted to more realistic circumstances regarding the responsibility of fishers. Alleviating the pressure on fishers and establishing more realistic objectives will improve their commitment.

To this end, the analysis is based on the estimation of General Additive Models (GAMs). This methodological approach pioneered by Hastie and Tibshirani (1990) enables to generalize a linear parametric model allowing for an additive nonlinear relationship between the response and each explanatory variable without the need of specifying the underlying smooth functional form. This estimation technique avoids trying different specifications, provides more precise predictions and offers testing for linear relations. It has been already used in fisheries estimation for the analysis of particular fisheries (Pulver and Stephen, 2019; Carbonell et al., 2018; Maina et al., 2018; Tsagarakis et al., 2014; Feekings et al., 2012). The resulting estimations of this analysis inform on the main determinants of discards, quantifying the relation and thus the impact of each variable on these catches.

The present analysis is structured as follows. Section 4.2 presents the data used in the analysis. Section 4.3 details the methods applied, describing the main features of the variables included in each model. Section 4.4 describes the main features of the variables included in the models and presents their estimation. Finally, Section 4.5 summarizes the most relevant discoveries of this analysis.

4.2 Description of the data

The data used in this analysis is extracted from multiple sources: information on the catches of countries, landings and discards is taken from the Sea Around Us project (Pauly and Zeller, 2015) and includes the quantity, value, species and harvesting area of catches; aquaculture production of countries, fish consumption and fish exports are extracted from The Food and Agricultural Organization of the United Nations (FAO, 2019a,b,c); the existence of discards regulation is documented in Condie et al. (2014); and the Gross Domestic Product of countries in constant 2010 US dollars is taken from the World Bank (World Bank, 2019). The sample used for this analysis contains between 65 and 137 countries for the period 1961 to 2016.

4.3 Methodology

The present analysis contributes to the design of international policies to minimize discards by showing the main socioeconomic and technological determinants of discarded catches. In particular, the analysis examines how the fishing activity of countries (i.e. the intensity, the fishing gears used, the species caught and the harvested areas), economic factors (i.e. the value of catches, the fish production alternatives and the industrialization of countries), social factors (i.e. the fish demand) and regulation (i.e. the existence of discarding bans) affect the discards of countries.

The specified General Additive Models are estimated using the mgcv package (Wood, 2017) in R (R Core Team, 2017) which includes an automatic data-driven smoothing selection. The smoothing level regulates the flexibility of the functional forms considered.

The starting model (Model 1) to determine discards explores all possible nonlinearities:

$$\begin{aligned}
\text{Discards}_{it} = & \text{s(Discards relative value)}_{it} + \text{s(Total landings)}_{it} \\
& + \text{s(Percentage of EEZ landings)}_{it} + \text{s(Aquaculture)}_{t} \\
& + \text{s(Fish consumption)}_{it} + \text{s(Fish exports)}_{it} \\
& + \text{Discards regulation}_{it} \\
& + \text{s(GDP)}_{it} + \text{s(year)}_{t} + \text{u}_{it},
\end{aligned}$$

where *Discards* are measured as annual tonnes of fish caught and returned to the water by countries in their fishing activity, subindex *it* refers to the observation of country i (i $=1,\dots,$N) in period t (t $=1,\dots,$T);

s(·) denotes a smooth, unknown and probably nonlinear function of the explanatory variable involved; and u_{it} accounts for the error term. Next we describe briefly the included explanatory variables.

The *Discards relative value* compares the monetary value assigned to discarded catches with the value of landings, which represents the monetary incentives of fishers to land the discarded catches. The net effect of this variable is determined through the interaction of demand and supply. From the supply side, the value of discards is negatively related to the amount of catches discarded. Indeed, the low market price of discards has been found to be one major reason for not landing them (Maynou et al., 2018; Tsagarakis et al., 2014). From the demand side, the value of discards may influence positively the quantity of discards through the decisions of consumers (Tsagarakis et al., 2014). On the one hand, higher price may induce consumers to replace the consumption of discarded fish by landed fish, reducing the incentives of fishers to land discarded catches. On the other hand, increases in the relative price of discards diminish the purchasing power of consumers, decreasing the market size and the incentives of fishers to land discarded catches.

Total landings accounts for the level of fishing activity by measuring the commercialized resources obtained from it. By definition of the variable, catches are positively associated with landings. Since discards are considered a proportion of catches (Pérez Roda et al., 2019; FAO, 2018), *Landings* is expected to have a positive effect on discards.

The *Percentage of EEZ landings* quantifies the dependence of fishing activity on resources from the Economic Exclusive Zone (EEZ) of the fishing entity. The UNCLOS regulation implemented by the UN (1982) defines the EEZ of a country as the waters within the 200-nautical miles from its coastline, establishing only free access to this area for the adjacent country. Beyond this limit, waters (denoted high seas) are considered of common property. This new status quo led to a clear expansion of fishing activities from the EEZs to the high seas. Over time, this movement caused high seas resources to be overexploited (Cullis-Suzuki and Pauly, 2010) although more evenly among countries (Gutiérrez and Inguanzo, 2019). *Percentage of EEZ landings* aims to reveal whether countries behave differently, in terms of discards production, depending on the ownership of the resources exploited. It might be expected that countries produce more discards on high seas as the shared cost of their generation may be perceived to be lower than the benefit obtained (Hardin, 1968). Additionally, the unwillingness to burden with the same responsibilities in unequal situations may induce them not to be so cautious when producing discards in high seas (Owusu et al., 2019; Fabinyi et al., 2015).

The commercialization of fishery resources allows countries to satisfy fish demand beyond national boundaries, fulfilling national (*Fish consumption*) and international (*Fish exports*) fish demand (FAO, 2018). The expansion of market size and economic incentives associated to larger fish demand decreases the amount of discarded catches (Van Putten et al., 2019). However, self-consumption of fished resources may have a larger effect on discards.

The *Aquaculture production* measures the annual tonnes of farmed fish produced globally. The stability of the farmed fish production and the development of aquaculture have furthered the presence of farmed fish in human fish consumption (FAO, 2018). Even though the production of farmed fish competes with the fishing industry to cover fish demand, aquaculture has been claimed as a complement of fishing to satisfy the increasing demand of fish that cannot be covered with the stagnating wild catches (Asche and Smith, 2018). Moreover, the aquaculture activity requires from wild fish to feed the farmed animals, expanding the market size and the economic revenues of otherwise discarded catches (Hasan and Halwart, 2009). Therefore, aquaculture production is expected to decrease discards.

Discards regulation is defined as a dummy variable that takes value 1 if there exists any kind of discarding regulation for the corresponding country and year. Given its nature, it is included linearly in the model. The implementation of discards regulation contributes positively to the reduction of discards altough it should be jointly implemented with other policies to improve the technological conditions of the fishing gears and increase the social awareness to ensure its success (Da-Rocha et al., 2018; Maynou et al., 2018; Raúl Prellezo, 2016; Gullestad et al., 2015; Condie et al., 2014; Johnsen and Eliasen, 2011). As there is not a common global standard application and measurement of discarding regulations, the broad existence of any discarding regulation has been considered for this analysis (Condie et al., 2014). Consequently, *Discards regulation* is expected to reflect the negative effect of these policies on discards.

The Gross Domestic Product (*GDP*) of countries, measured in 2010 constant US $, has been incorporated in the model to reflect that differences in the industrialization of countries may result in fishing techniques with dissimilar selectivity (Tsagarakis et al., 2014). Tsagarakis et al. (2014) find for Mediterranean fisheries a positive relationship between the GDP of the countries and their discard ratio. This result seems reasonable as wealthier countries own more resources, becoming more selective with the fish consumed. On the contrary, poorer countries are associated with lower discards due to the larger nutritional dependence on these resources. However, it should be borne in mind that when societies become richer, they become aware of the negative externalities generated by the growth process and begin to implement policies aimed at reducing these negative effects. This idea agrees with the Environmental Kuznets Curve (EKC) that, based on Kuznets (1955), hypothesizes an inverted U-shaped relationship between indicators of environmental degradation and income. Discards can be seen as one of these negative externalities. Analyzing discards at the country level makes it possible to contrast the validity of EKC as well as estimating the inflection point beyond which discards begin to be reduced despite economic growth.

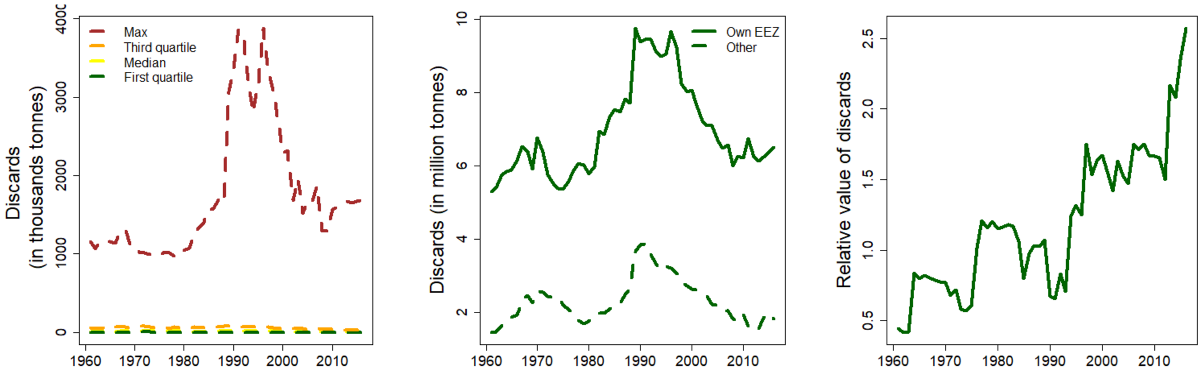
Finally a smooth trend is also included, *s(year)*, to account for the influence of some common factors influencing overall discards that vary over time and are not captured by the remaining explanatory variables. Such common factors can contemplate, for instance, perturbations in the conditions of the ecosystems due to climate change (Gaines et al., 2018; Tu et al., 2018; Lam et al., 2016; Breitburg, 2002).

4.4 Results

This Section presents the main features of the variables used in the analysis and the results obtained in the estimations of the GAMs described previously.

4.4.1 Descriptive analysis

Global discards are estimated to be over 8 million tonnes in 2016, doubling the estimated quantity for 1950. Discarded catches are heterogeneously distributed between countries as shown in Figure 4.1(a). The major gap in discards appears between the countries in the fourth quartile (i.e. 25% of countries with the largest discards) and the remaining ones. In 2016, each country in the fourth quartile produced between 32 and 1690 thousand tonnes. In particular, around 45% of the global discards in 2016 were produced by 4 countries: the Russian Federation, with over 20% of discards; China, with almost 10% of discards; USA, with more than 8% and Vietnam, with around 7%. Despite the largest dissimilarity in discards is found in the fourth quartile,



(a) Time trends

(b) Discards by areas

(c) Relative value trend

Figure 4.1: Discards distribution.

differences between the remaining quartiles are also noticeable. In 2016, countries in the first quartile (i.e. 25% of countries with the lowest discards) produced less than 1 thousand tonnes each, countries in the second quartile (i.e. 50% of countries with the lowest discards) produced less than 9 thousand tonnes each and countries in the third quartile (i.e. 75% of countries with the lowest discards) produced less than 32 thousand tonnes each.

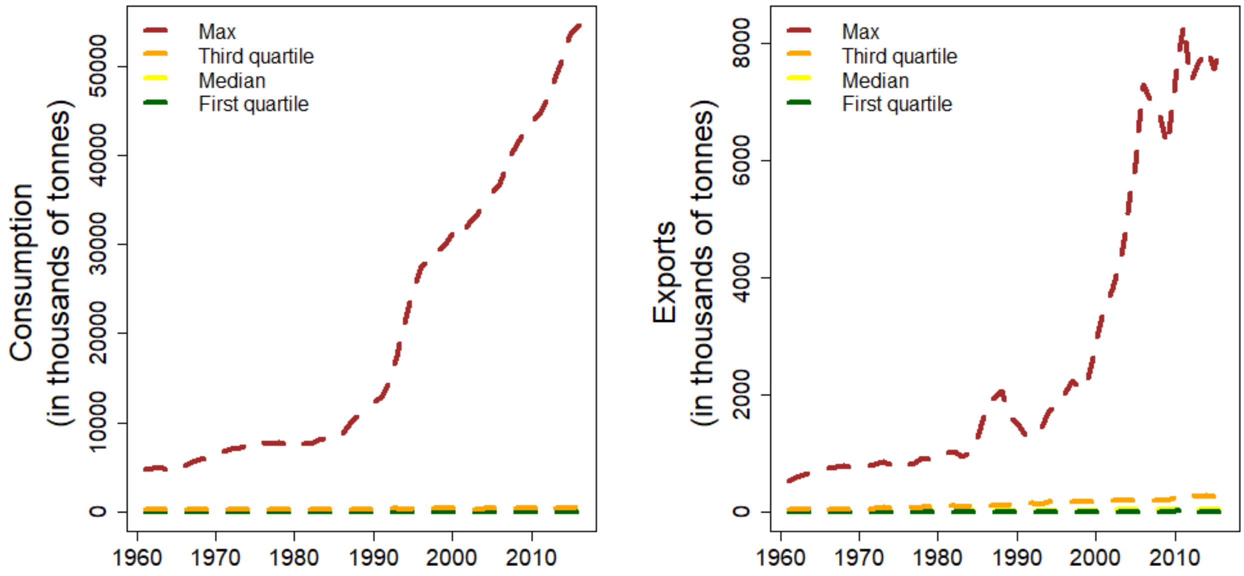
By areas, discards are mostly produced within the EEZ of countries even after the implementation of the UNCLOS (Figure 4.1(b)). The continuous larger generation of discards within private fishing areas justifies the concerns emerged regarding the effectiveness of the UNCLOS in maintaining the exploitation under sustainable limits in those fishing areas (Pauly et al., 2002; Jackson et al., 2001).

On average, the value of each ton discarded is larger than the value of each ton landed for most of the period under analysis (Figure 4.1(c)). In particular, the value of each ton discarded was more than twice the value of each ton landed in 2016. Since the monetary value of catches plays a major role in discarding decisions, the large relative value of discards may reflect the insufficient demand for those catches.

Fish demand, represented by fish consumption and exports, varies significantly across countries (Figure 4.2). The largest heterogeneity is found in the fourth quartile (i.e. 25% of countries with largest consumption or exports). In 2016, countries in the fourth quartile consumed between 0.5 and 55 million tonnes and exported between 0.2 and 8 million tonnes. At a lower extent, heterogeneity in the consumption and exports of the other quartiles is also remarkable. With respect to the distribution of consumption in 2016: the first quartile (i.e. 25% of countries with the lowest consumption) demanded less than 24 thousand tonnes; the second quartile (i.e. 50% of countries with the lowest consumption) demanded up to 115 thousand tonnes and the third quartile (i.e. 75% of countries with the lowest consumption) demanded up to 0.5 million tonnes. Regarding the distribution of exports in 2016: the first quartile (i.e. 25% of countries with the lowest exports) exported up to 6 thousand tonnes; the second quartile (i.e. 50% of countries with the lowest exports) exported less than 50 thousand tonnes and the third quartile (i.e. 75% of countries with the lowest exports) exported up to 250 thousand tonnes.

Aquaculture production has increased in more than 106 million tonnes from 1961 to 2016 (Figure 4.3), being the Asian continent the largest producer (FAO, 2018).

The distribution of landings according to gear types and species are shown in Figure 4.4. Due to the large number of gear types and species found in the data, annual quartiles have been applied to gather them. Fishing gears are grouped according to their inefficiency, measured by the ratio discards/catches, aiming to reflect



(a) Fish consumption

(b) Fish exports

Figure 4.2: Distribution of fish demand.

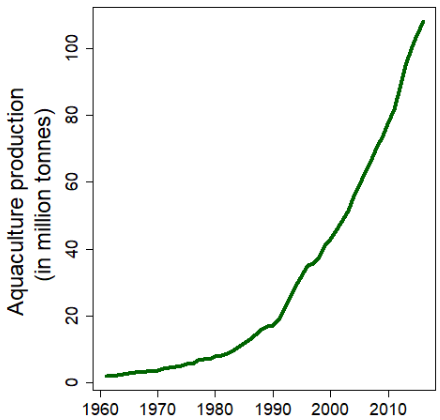
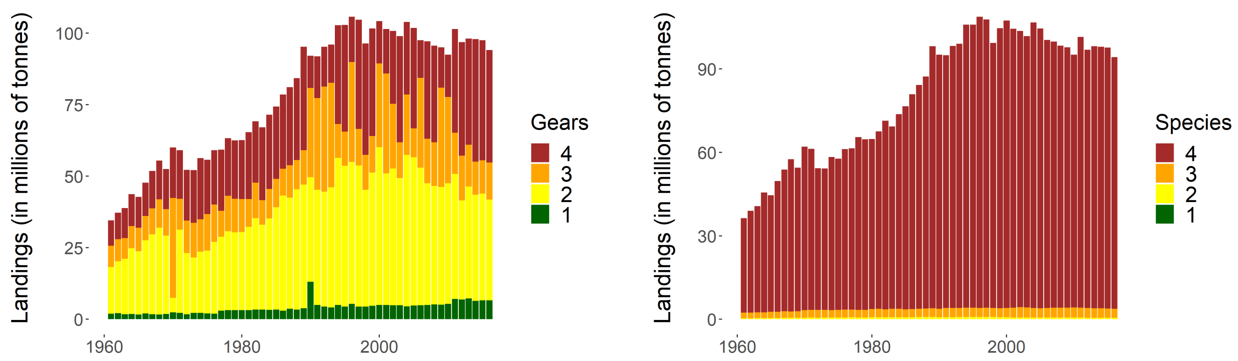


Figure 4.3: Aquaculture production.



(a) Landings by gear

(b) Landings by species

Figure 4.4: Distribution of landings.

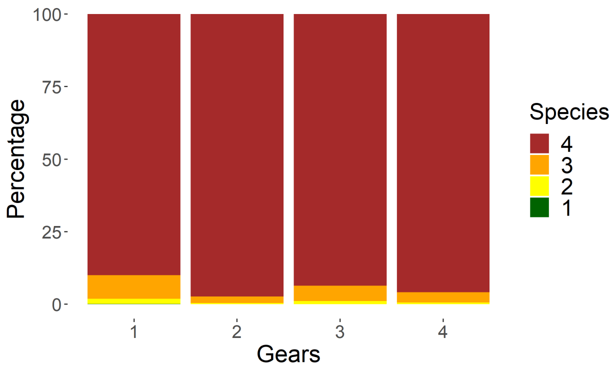


Figure 4.5: Species landed by gears.

their heterogeneous selectivity (Cashion et al., 2018). In particular, higher quartiles refer to more inefficient gear types, producing more discards per catches. See Appendix 4.A for the classification of gear types in quartiles. Species are classified according to their monetary value in order to account for the economic incentive of fishers to land more valuable catches (Maynou et al., 2018; Tsagarakis et al., 2014; Feekings et al., 2013). This classification places more valued species in higher quartiles. See Appendix 4.B for the classification of species in quartiles.

According to above classification, it can be observed that fishing activity differs in terms of resources obtained as shown in Figure 4.4(a). Gears in the second quartile (almost the most efficient in terms of discards per catches) such as purse seines, hands or tools and small scale encircling nets are responsible for most of the landings from 1961 to 2016. However, the contribution to landings from gears in the fourth quartile (the most inefficient in terms of discards per catches) such as shrimp trawls, beam trawls, otter trawls and bottom trawls has increased noticeably over the period under analysis. Indeed, in 2016, the landings from gears in the fourth quartile (over 39 million tonnes) exceeded the landings from those in the second quartile (over 35 million tonnes). Landings also vary between species (Figure 4.4(b)). Species in the fourth quartile (most valued species) such as *Engraulis ringens, Theragra chalcogramma* and *Sardinops sagax* are the most landed. Figure 4.5 shows that gears from every quartile land mostly species in the fourth and third quartiles.

4.4.2 Determinants of discards

The estimation of Model 1 is presented in Table 4.1, being the smoothed terms of this model illustrated in Figure 4.6. Table 4.1 displays results differently depending on how variables are included in the model. If variables are included as linear terms, the point-wise estimation of the coefficients and the standard deviation, in parentheses, are provided. Nonetheless, if variables are included as smooth terms, the effective degrees of freedom (edf) are shown. Edf are related with the smoothness level and, consequently, with the level of non-linearity of the functional form. Values of edf close to zero mean that the variable is not significant; values equal to 1 reflect linear relations; values between 1 and 2 show weak nonlinear relations; and values larger than 2 imply high non-linear relations (Hunsicker et al., 2016; Zuur et al., 2009). In both cases, linear and non-linear terms, asterisks indicate their statistical significance. All estimations are obtained by assuming a Gaussian family and an identity link function.

Table 4.1 reveals three major types of relationships between the explanatory variables considered in Model

Table 4.1: Socioeconomic determinants of discards.

	Dependent variable: Discards	
Model	(1)	(2)
Parametric terms		
Discards Regulation	84,838.600*** (19,073.690)	−67,493.340*** (13,935.140)
Landings 11	-	−8.205 (8.318)
Landings 12	-	−3.762** (1.572)
Landings 13	-	−1.901*** (0.357)
Landings 14	-	0.197*** (0.017)
Landings 21	-	−4.334 (9.947)
Landings 22	-	3.525*** (1.046)
Landings 23	-	−0.458** (0.213)
Landings 24	-	0.036*** (0.002)
Landings 31	-	37.038*** (10.354)
Landings 32	-	1.725** (0.852)
Landings 33	-	1.023*** (0.192)
Landings 34	-	0.156*** (0.006)
Landings 41	-	25.195** (10.003)
Landings 42	-	2.219** (0.985)
Landings 43	-	−1.202*** (0.205)
Landings 44	-	0.385*** (0.006)
Non-Parametric terms		
s(Consumption)	14.021***	17.701***
s(Exports)	9.649***	15.878***
s(% EEZ landings)	9.173***	14.250***
s(GDP)	12.606***	16.777***
s(Year)	0.987***	5.452***
s(Aquaculture)	0.003	-
s(Relative price)	0.003	-
s(Landings)	15.309***	-
Observations	6,129	6,129
Adjusted R^2	0.593	0.746
AIC	165444.8	161009.8

Note: Statistical significance: * for p<0.1; ** for p<0.05 and *** for p<0.01.

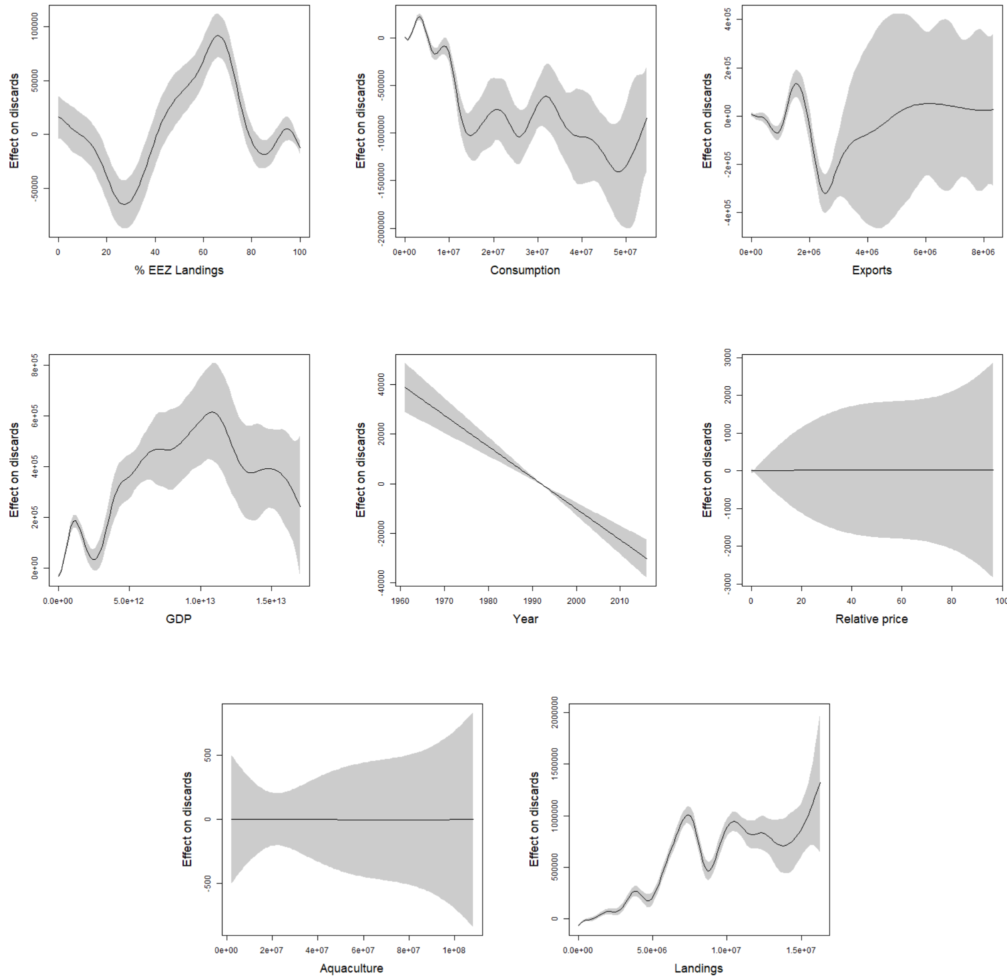


Figure 4.6: Model 1: smoothed terms.

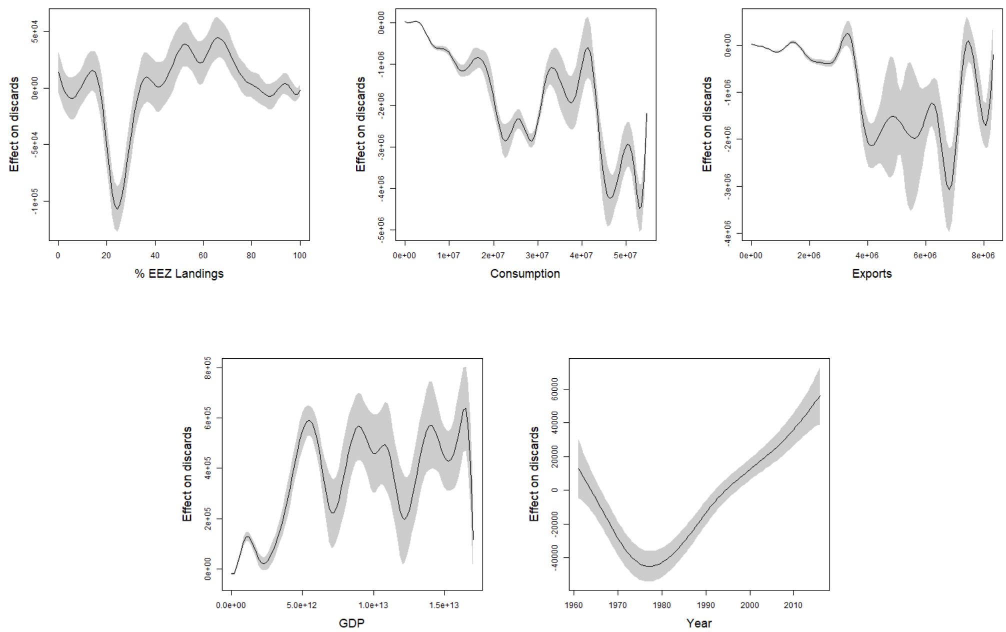


Figure 4.7: Model 2: smoothed terms.

1 and discards. In order to interpret the impact of each variable, it is assumed that the other factors remain constant (*ceteris paribus*). Firstly, *Discards relative value* and *Aquaculture production* seem not statistically significant when explaining discards. Since *Discards relative value* and *Aquaculture production* may be connected to other explanatory variables in the model, their effect on discards may be already represented by the effect of the other related explanatory variables. In the case of *Discards relative value*, jointly considering the market demand (*Consumption* and *Exports*) and supply (*Landings*) may already represent the monetary consequences of their interaction (*Discards relative value*). Regarding the *Aquaculture production*, this variable represents a small source of fish demand (as *Consumption* and *Exports*) and a supplier of fish (together with *Landings*). Being related to both sides of the market, the effect of *Aquaculture* may be included through *Consumption*, *Exports* and *Landings*. Secondly, the evolution of time seems to have a negative linear effect on discards as the *Year* edf is close to 1. By contrast, the implementation of discarding regulations seems to have a positive linear effect on discards. Thirdly, nonlinear relationships are observed between the remaining variables and discards. In general, fish demand reduces the catches discarded as expected by the increase in the economic incentives associated to the expansion of the market. However, discards seem to increase with some levels of demand, reflecting the selectivity associated to the demand of particular species and the need to familiarize consumers with alternative fishes. Fishing in private areas does not have a linear effect either. For medium levels of dependence on EEZ resources, the effect on discards increases. This is in line with Jackson et al. (2001) and Pauly et al. (2002), who question the historical efficacy of the UNCLOS regulation on ensuring the viability of coastal areas. The positive effect disappears for high dependence on these resources. When most of fishery resources come from EEZ, fishing activity is associated with lower discards. As observed in Tsagarakis et al. (2014), the effect of GDP on discards is nonlinear. Indeed, it describes several Kuznets curves corresponding to different levels of industrialization. Each of these curves reflects that the fishing activity associated with more industrialization generates more discards until a further level of industrialization is reach, allowing to develop and apply more efficient fishing techniques. Regarding the fishing activity, discards increase with the level of *Landings*. Nevertheless, such increase is not constant, raising objections to the representation of discards as a fixed proportion of catches and highlighting the existence of additional factors in the fishing activity that determine the amount of discarded catches.

Aiming to account for the factors producing the nonlinear effect of landings and the findings from the estimation of Model 1, the following model (Model 2) has been defined and estimated:

$$
\begin{aligned}
\text{Discards}_{it} = &\sum_{l=1}^{16} \text{Landings gears and species}_{lit} \\
&+ \text{Discards regulation}_{it} \\
&+ \text{s(Percentage of EEZ landings)}_{it} \\
&+ \text{s(Fish consumption)}_{it} + \text{s(Fish exports)}_{it} \\
&+ \text{s(GDP)}_{it} + \text{s(year)}_{t} + \text{u}_{it},
\end{aligned}
$$

where *Landings gears and species* refer to the landings of each species group coming from each gear type and the sub-index $l = (1, \ldots, 16)$ denotes each particular combination of gear and species groups. Fishing gears are grouped in quartiles according to their inefficiency measured as the ratio of discards per catches (see Appendix 4.A). Since higher quartiles gather more inefficient gears, these groups are expected to have larger effect on discards (Cashion et al., 2018). Species are classified in quartiles according to their monetary value (see Appendix 4.B). Given that more valued species are placed on higher quartiles, these are expected to have a smaller impact on discards (Maynou et al., 2018; Tsagarakis et al., 2014; Feekings et al., 2013). Once gear types and species are classified into quartiles, the fishing activity of countries is aggregated to measure the landings coming from each group of gears and species. As a result, their union gives rise to sixteen groups. Following the hypotheses for its components, the effect of *Landings gears and species* in discards is expected to be positive and larger for groups with more inefficient gears or less valued species. Table 4.1 presents the estimation of this model and Figure 4.7. In this table, $Landings_{gm}$ correspond to the landings of gear quartile g and species quartile m.

Omitting the non-significant variables and specifying landings linearly as a combination of the gear used and the species caught has improved the performance of the model in terms of the Akaike's Information Criteria (AIC, Hastie and Tibshirani (1990)). The AIC measures the likelihood of the model penalizing for the number of variables used in its specification. Thus, it allows to compare the capability of different models to fit data while accounting for the number of variables devoted to this purpose. In particular, lower values of the AIC indicate that models fit better the data. Having an AIC below Model 1, Model 2 is found to represent better the data.

Table 4.1 and Figure 4.7 show significant variations with respect to the estimations of Model 1. Firstly, the regulations unilaterally enacted by certain countries during the period analyzed to minimize the production of discards seem effective. This success motivates the design of international policies for the minimization of discards keeping in mind the experience of countries that have already implemented discarding regulations. Secondly, the negative impact of fish demand becomes clearer for most of the consumption and exports ranges. Thirdly, the relationship between discards and the dependence of EEZ resources has accentuated for low levels of EEZ dependence and soothed for high levels of EEZ dependence. However, the pattern remains similar. Low levels of dependence on these resources are associated with a more efficient exploitation, increasing discards with a larger dependence on them. Fourthly, the Kuznets curves described in the GDP effect have intensified. In particular, the effect of this variable on discards describes five Kuznets curves with turning points on 0.9, 5.3, 8.8, 13.9 and 16.3 billion 2010 constant US dollars. Fifthly, the evolution of *year* has become non-linear. This non-linearity reveals that discards experienced a downward trend from 1960 to 1980, approximately. Around 1980, this trend reversed and the impact of global factors on discards increased, over-passing the initial level of discards in 2016. Finally, the segmentation of landings by fishing gears and species confirms that the activity of more inefficient gears (like bottom trawls, shrimp trawls, otter trawls, beam trawls, dragged gears, gillnets, lines,...) generates more discards than the activity of more efficient (artisanal) fishing gears. The magnitude of their effect varies with the value of species. In this respect, this model reveals that each group of gears performs better with species of different value in terms of discards. Regarding their efficiency, specialization of gears on different groups of species could be proposed in order to minimize discards. In particular, gears in the first quartile (bagnets, harpoons, pole and lines,...) seem the most efficient in species from the first to the third quartiles whereas gears in the second quartile (purse seines, hand or tools, small scale encircling nets,...) seem the most efficient in most valued species. According to their levels of efficiency between the different groups of species, gears in the third quartile (gillnets, lines,...) perform better regarding most valued species while gears in the fourth quartile (bottom trawls, shrimp trawls, otter trawls, beam trawls, dragged gears,...) discard less with species in the third quartile. Given that this specialization could incentivize fishers not to improve their efficiency due to their advantageous position in fishing more lucrative species, the improvement of their overall efficiency should be encouraged. Special effort must be placed in increasing the efficiency of gears in the third and fourth quartiles, which are the largest producers of discards concerning less valued species.

4.5 Conclusions

Discards refer to the catches taken in the fishing activity and returned to the waters, usually dead or highly damaged in this process (Feekings et al., 2012). These catches are considered a waste from the point of view of fishing activity, in both economic and environmental terms. In economic terms, discards entangle significant private and social costs. From the private perspective, discards imply a loss of profits due to the unproductive use of fishing production factors (Clucas, 1997). From the social perspective, discards threaten the food security by reducing the availability of potential food (FAO, 2018; Diamond and Beukers-Stewart, 2011; Blanco et al., 2007). In environmental terms, discards perturb the multi-species balance of the ecosystems (Clucas, 1997).

Between 1950 and 2016 discards are estimated to have almost doubled, increasing from 4.34 to 8.35 million tonnes. Discards are unevenly produced by countries. In 2016, the largest producers were the Russian Federation (representing around 20% of global discards), China (accounting for 10% of global discards), the USA (with more than 8% of global discards) and Vietnam (producing around 7% of global discards) (Pauly and Zeller, 2015).

The reasons for discarding catches are of diverse nature. Theoretical analyses have modeled discards as the rational consequence of cost-benefit comparisons and capacity constraints (Pascoe, 1997; Vestergaard, 1996; Anderson, 1994; Arnason, 1994). Discards have been also theoretically modeled as the consequence of weaknesses in some regulatory frameworks for managing fisheries (Singh and Weninger, 2015; Hatcher, 2014; Holland, 2010; Abbott and Wilen, 2009; Herrera, 2005; Turner, 1997). Empirical analyses have explained discards of vessels through the technical features of the vessel and environmental variables from the fishing area (Bellido et al., 2019; Fauconnet et al., 2019; Pulver and Stephen, 2019; Carbonell et al., 2018; Madsen et al., 2018; Maina et al., 2018; Pointin et al., 2018; Maeda et al., 2017; Pennino et al., 2014; Tsagarakis et al., 2014; Feekings et al., 2013, 2012). Across literature, the most frequent reasons attributed to discards are the technological inefficiency and the low economic incentive for landing discarded catches. Indeed, the variation in discards across gear types is significant. While bottom trawlers discard up to 20% of their catches, purse seiners, the largest producers of landings, discard 4% of their catches (Pérez Roda et al., 2019; Cashion et al., 2018). Regarding the value of catches, landings are composed of the most lucrative species (Pauly and Zeller, 2015).

Sustainable fishing practices for the conservation of the marine environment and food security have been globally recognized as priority issues (FAO, 2018; UN, 2015; CBD, 2010). Nevertheless, international policies on discarding, process compromising both of these aspects, have not been considered yet. Unilaterally, certain regions have already implemented discarding bans mixing technological and market regulations (Condie et al., 2014).

The present analysis contributes to the design of feasible and efficient international discarding policies by showing the main technological, socioeconomic and environmental factors underlying the global discards produced between 1961 and 2016. The combination of these factors provides more accurate estimations of their impact on discards, allowing to fit policies according to real circumstances. Technological factors are represented through the fishing activity comparison of the gear types. Socioeconomic factors involve the commercial value of catches, the ownership of the exploited resources, the industrialization of the country and the existence of alternatives in fish production. Changes in the environmental conditions are included in a time-varying trend.

For this purpose, a sequence of General Additive Models (GAMs) (Hastie and Tibshirani, 1990) is implemented. The initial model (Model 1) allows all variables to be included in nonlinear terms. The second model (Model 2) is adapted following evidences from the Model 1. Specifically, Model 2 omits non-significant variables in Model 1 and decomposes linearly the effect of landings considering the technological differences between gear types and the unequal economic incentives of differently valued species. The estimation of Model 2 is the best ranked in terms of Akaike's Information Criteria and Adjusted R^2. In particular, this model explains almost 75% of the global discards from 1961 to 2016. Due to the availability of information for some of the variables considered in these models, the sample used for this analysis contains information for 65 to 137 countries from 1961 to 2016.

Numerous findings can be highlighted from the estimations of the analysis. Firstly, the discarding bans already implemented in British Columbia, Faroe Island, Iceland, New Zealand, Norway and US Alaska seem successful in decreasing discards. Consequently, the design of international regulations to minimize discards should consider the previous experiences of these regions. Secondly, fish demand (including fish consumption and fish exports) contribute to decrease discards. However, its effect varies with its levels. In general, the negative effect on discards increases with more tonnes consumed and exported. Thus, the promotion of fish demand could help to decrease discards. Thirdly, discards are differently affected by gear types and species. Small scale gillnets, lines, gillnets, otter trawls, beam trawls, shrimp trawls, dredges and bottom trawls are among the most inefficient gears, producing more discards per ton of catches. Special effort should be made in improving the selectivity of these gears. The effect of gears on discards also varies with the value of species. Artisanal gears (harpoon, bagnets, pole and line,...) are more efficient in less valued species. Hand or tools, purse seines and small scale encircling nets performed better in terms of discards for most valued species. Small scale gillnets, lines and gillnets produce less discards of most valued species in comparison to other groups of species. Similarly, otter trawls, beam trawls, shrimp trawls, dredges and bottom trawls are more efficient in most valued species in comparison to other groups of species. Therefore, the fishing activity of the different gears could be oriented towards the species for which they are the most efficient. Given the lucrative benefit from specializing in most valued species, gears already efficient in these groups of species could be tempted not to improve their efficiency fishing other species. Consequently, overall selectivity of gears should be encouraged. Fourthly, the industrialization of countries has a nonlinear effect on discards, describing multiple Environmental Kuznets Curves (EKC) through the different levels of industrialization. Within each EKC, initial increases in GDP enlarge the production of discards, but this effect reverses for further increases. Fifthly, the evolution of the time-trend reflects the existence of external factors affecting global discards. The effect of these factors decreased from 1961 to the late 1970s'. Since then, the effect kept increasing and became positive in the 1990s'.

The evidences from this analysis suggest that international policies for the minimization of discards should encourage fish demand and promote the selectivity of gear types, especially small scale gillnets, lines, gillnets, otter trawls, beam trawls, shrimp trawls, dredges and bottom trawls. In addition, gear types could be oriented to fish the species for which they are more efficient. Finally, results from this analysis show that the previous experiences of the regions with discarding bans implemented should not be disregarded.

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Appendix 4.A Classification of gear types in quartiles

The following Table shows the gears included in each quartile of the efficiency distribution (*Cluster*). *Occurrences* indicates the frequency (in percentage terms) with which gears are included in the quartile in the period under analysis. *Landings* quantifies the value of catches (in 2010 constant US million $) landed by the corresponding gear during the period under analysis. Information on gear types appearing in a cluster less than 50% of the times in the period under analysis is omitted for clarity purposes.

Table 4.A.1: Classification of gear types.

Gear	Cluster	Occurrences	Landings
harpoon	1	100	9893862609.77951
unknown by author	1	100	128594511144.745
recreational fishing gear	1	100	2121713990614.9
bagnets	1	100	12830137835522.9
pole and line	1	97.01	3788329223370.37
cast nets	1	91.04	2206936033189.62
pots or traps	1	76.12	3423672433279.23
small scale trammel net	1	74.63	35156853958.2719
hand or tools	2	97.01	10872748672608.2
purse seine	2	97.01	3567609436123240
small scale encircling nets	2	86.57	10365211446013.3
small scale hand lines	2	77.61	3998261821070.76
small scale lines	2	68.66	14864872526814
small scale seine nets	2	61.19	9917050406187.38
artisanal fishing gear	2	59.70	13750530394029.5
other	2	53.73	4471837460489.61
small scale pots or traps	2	53.73	16945651165008
small scale gillnets	3	92.54	43203214721096.2
lines	3	86.57	1149705223201.1
gillnet	3	83.58	43417670201428.8
small scale longline	3	82.09	2863681785465.65
small scale purse seine	3	68.66	23164517312252.8
pelagic trawl	3	52.24	308382680843170

Gear	Cluster	Occurrences	Landings
other nets	3	50.75	8509651125633.89
otter trawl	4	100	34892458036.8849
beam trawl	4	100	74449915089.5755
mixed gear	4	100	3158276876107.54
shrimp trawl	4	100	3848087874141.03
dredge	4	89.55	7177101119665.45
bottom trawl	4	85.07	490055163661073
small scale other nets	4	82.09	4535037167214.62
dragged gear	4	67.16	1278517816.02592
longline	4	53.73	21827495223912.5

Appendix 4.B Classification of species in quartiles

The following Table shows the species included in each quartile of the value distribution (*Cluster*). *Occurrences* indicates the frequency (in percentage terms) with which species are included in the quartile in the period under analysis. *Landings* quantifies the value of catches (in 2010 constant US million $) from each species landed during the period under analysis. Information on species appearing in a cluster less than 80% of the times in the period under analysis is omitted for clarity purposes.

Table 4.B.1: Classification of species.

Species	Cluster	Occurrences	Landings
Kyphosus	1	100	30868415.22
Chaetodontidae	1	100	2910722.64
Carcharhinus albimarginatus	1	100	2867350.8
Mulloidichthys flavolineatus	1	100	2589489.66
Scarus rubroviolaceus	1	100	1615296.06
Lethrinus rubrioperculatus	1	100	1471387.5
Etelis coruscans	1	100	1336712.2
Lutjanus apodus	1	100	1264139.97
Hyporthodus mystacinus	1	100	1243419.47
Epinephelus merra	1	100	1194912.94
Epinephelus chlorostigma	1	100	1061038.88
Pseudopentaceros richardsoni	1	100	892880.68
Rhizoprionodon porosus	1	100	852820.19
Parupeneus	1	100	849212.74
Etelis carbunculus	1	100	802590.73
Polyprion oxygeneios	1	100	799360.38
Sphyraena novaehollandiae	1	100	744286.12

Species	Cluster	Occurrences	Landings
Siganus lineatus	1	100	740289.86
Myripristis	1	100	729108.19
Cephalopholis	1	100	670104.62
Echinozoa	1	100	661252.62
Acanthurus triostegus	1	100	647981.01
Lutjanus russellii	1	100	531426.48
Pristidae	1	100	460721.34
Cephalopholis cruentata	1	100	374842.63
Nebrius ferrugineus	1	100	352788.85
Synanceiidae	1	100	333708.65
Terebridae	1	100	322653.3
Larimus acclivis	1	100	282586.41
Lethrinus variegatus	1	100	256610.28
Gymnocranius euanus	1	100	255991.66
Pomadasys panamensis	1	100	227343.68
Haemulon flavolineatum	1	100	209664.51
Archosargus rhomboidalis	1	100	197441.06
Panulirus regius	1	100	195947.88
Balistes vetula	1	100	174487.41
Lethrinus atkinsoni	1	100	167364.21
Platax orbicularis	1	100	160977.67
Cheilinus trilobatus	1	100	150526.26
Diapterus peruvianus	1	100	136554.32
Uranoscopus	1	100	130799.47
Calamus calamus	1	100	124484.09
Gymnothorax moringa	1	100	120866.98
Latris lineata	1	100	110962.35
Cephalopholis argus	1	100	105854.73
Canthidermis sufflamen	1	100	104243.91
Mulloidichthys	1	100	102649.07
Kyphosus bigibbus	1	100	98261.59
Lutjanus fulviflamma	1	100	89713.91
Myripristis adusta	1	100	74998.21
Scarus oviceps	1	100	72299.53
Tridacnidae	1	100	68018.51
Gerres cinereus	1	100	62386.44
Haemulon album	1	100	57081.04
Tresus	1	100	54296.09
Dasyatis guttata	1	100	54133.79

Species	Cluster	Occurrences	Landings
Aethaloperca rogaa	1	100	51027.46
Aphareus furca	1	100	50819.07
Portunus sanguinolentus	1	100	49959.26
Tenualosa toli	1	100	44956.95
Neoniphon marianus	1	100	42007.01
Uraspis secunda	1	100	40808.33
Cantherhines	1	100	36575.76
Trachurus novaezelandiae	1	100	35736.67
Caulolatilus chrysops	1	100	33824.75
Kyphosus vaigiensis	1	100	31341.94
Sparisoma chrysopterum	1	100	30773.8
Gnathodentex aureolineatus	1	100	28703
Aluterus monoceros	1	100	25947.8
Cheilinus	1	100	25324.08
Abudefduf vaigiensis	1	100	24317.33
Epinephelus morrhua	1	100	23449.31
Acanthurus blochii	1	100	23383.27
Paranthias furcifer	1	100	23135.96
Holocentrus rufus	1	100	22618.12
Strombus luhuanus	1	100	21616.09
Epinephelus summana	1	100	21279.27
Coregonus pidschian	1	100	19865.69
Plectorhinchus pictus	1	100	19547.97
Parribacus caledonicus	1	100	19488.42
Epinephelus lanceolatus	1	100	18940.22
Sparisoma rubripinne	1	100	18928.33
Pomadasys corvinaeformis	1	100	17842.98
Scarus coeruleus	1	100	17441.26
Enteroctopus dofleini	1	100	15740.94
Pristipomoides sieboldii	1	100	15447.15
Parupeneus cyclostomus	1	100	14517.62
Bothus mancus	1	100	12140.56
Lampris	1	100	12131.57
Myripristis violacea	1	100	11819.4
Hyporhamphus ihi	1	100	10263.29
Myoxocephalus quadricornis	1	100	8523.56
Variola albimarginata	1	100	6710.04
Naso annulatus	1	100	5638.58
Lambis lambis	1	100	5244.68

Species	Cluster	Occurrences	Landings
Pomacanthidae	1	100	4769.66
Mycteroperca interstitialis	1	100	4657.47
Lutjanus monostigma	1	100	4291.43
Coregonus laurettae	1	100	3869.3
Mycteroperca tigris	1	100	3732.34
Calotomus carolinus	1	100	3318.88
Trachinotus goodei	1	100	3258.86
Paracirrhites hemistictus	1	100	2235.66
Cirrhitus pinnulatus	1	100	2191.61
Upeneus taeniopterus	1	100	2013.45
Helicolenus mouchezi	1	100	1964.74
Sebastes capensis	1	100	1964.74
Trachurus longimanus	1	100	1964.74
Acanthurus bahianus	1	100	1854.77
Lethrinus amboinensis	1	100	1580.63
Torpedo	1	100	1537.78
Pristipomoides zonatus	1	100	1358.22
Epinephelus fasciatus	1	100	1304.83
Cephalopholis urodeta	1	100	1290.85
Hemiramphus archipelagicus	1	100	1182.7
Cheilinus undulatus	1	100	961.31
Monodactylus argenteus	1	100	567.74
Diodon hystrix	1	100	524.9
Wattsia mossambica	1	100	492.12
Ellochelon vaigiensis	1	100	458.43
Diplodus sargus helenae	1	100	432.67
Platax	1	100	374.63
Labrus	1	100	345.7
Myripristis vittata	1	100	339.83
Cittarium pica	1	100	188.64
Cephalopholis sonnerati	1	100	166.04
Cirrhitidae	1	100	123.58
Kuhliidae	1	100	102.44
Naso hexacanthus	1	100	94.81
Uraspis helvola	1	100	79.27
Lambis truncata	1	100	59.48
Aulostomus chinensis	1	100	46.65
Naso caesius	1	100	39.96
Epinephelus coeruleopunctatus	1	100	37.31

Species	Cluster	Occurrences	Landings
Labrus merula	1	100	35.85
Naso vlamingii	1	100	34.45
Myripristis amaena	1	100	26.79
Scarus dimidiatus	1	100	17
Caesio teres	1	100	15.29
Pomadasys maculatus	1	100	14.44
Chlorurus sordidus	1	100	12.29
Plectorhinchus gibbosus	1	100	8.58
Caranx papuensis	1	100	7.92
Myripristis chryseres	1	100	5.04
Myripristis woodsi	1	100	2.95
Abudefduf sordidus	1	100	1.21
Eriphia sebana	1	100	0.76
Myripristis kuntee	1	100	0.61
Corniger spinosus	1	100	0.14
Cephalopholis sexmaculata	1	100	0
Exocoetus volitans	1	100	0
Pterocaesio tile	1	100	0
Lutjanus kasmira	1	98.51	22230530.15
Calamus bajonado	1	98.51	734318.04
Epinephelus fuscoguttatus	1	98.51	683532.04
Tellinidae	1	98.51	557441.79
Coregonus nasus	1	98.51	540222.44
Manta birostris	1	98.51	447505.67
Kyphosus sectatrix	1	98.51	326358.7
Naso brevirostris	1	98.51	321159.74
Turbo marmoratus	1	98.51	248905.32
Herklotsichthys quadrimaculatus	1	98.51	175584.54
Hippocampus	1	98.51	146778.09
Aldrichetta forsteri	1	98.51	95225.27
Sphyraena putnamae	1	98.51	93574
Trachinotus cayennensis	1	98.51	69939.02
Trachinotus blochii	1	98.51	54964.93
Acanthurus sohal	1	98.51	52823.53
Pristipomoides flavipinnis	1	98.51	3320.76
Pentaceros decacanthus	1	98.51	3240.38
Mulloidichthys pfluegeri	1	98.51	3106.77
Sargocentron	1	98.51	2020.87
Saccostrea cucculata	1	98.51	1271.16

Species	Cluster	Occurrences	Landings
Mulloidichthys vanicolensis	1	97.01	4109701.01
Gerres longirostris	1	97.01	799080.5
Trachinotus mookalee	1	97.01	643915.21
Pomadasys argyreus	1	97.01	568916.55
Spratelloides	1	97.01	289975.95
Alosa alosa	1	97.01	86889.47
Pomatomidae	1	97.01	10645.31
Albula vulpes	1	95.52	4405106.37
Carcharhinus brachyurus	1	95.52	2294968.69
Carangoides fulvoguttatus	1	95.52	510136.41
Mola	1	95.52	499090.24
Fistulariidae	1	95.52	274677.04
Aphia minuta	1	95.52	139031.47
Sphyraena picudilla	1	95.52	58128.55
Neomyxus leuciscus	1	95.52	11434.45
Oxycheilinus unifasciatus	1	95.52	0.63
Panulirus gracilis	1	94.03	4636537.01
Cyttus traversi	1	94.03	3983020.07
Epinephelus labriformis	1	94.03	2242189.84
Alosa aestivalis	1	94.03	916183.7
Tridacna squamosa	1	94.03	709531.83
Holothuria atra	1	94.03	314617.45
Turbo setosus	1	94.03	192104.63
Torpedo marmorata	1	94.03	172942.12
Conger oceanicus	1	94.03	152051.38
Haemulon vittatum	1	94.03	140131.61
Fundulus majalis	1	94.03	30404.29
Heptranchias perlo	1	94.03	1370.15
Echinorhinus brucus	1	94.03	588.22
Pristipomoides auricilla	1	94.03	6.5
Caranx lugubris	1	92.54	27077971.23
Myripristis jacobus	1	92.54	5678213.6
Panulirus versicolor	1	92.54	3393337.99
Scarus persicus	1	92.54	834001.63
Aristeidae	1	92.54	786046.73
Fistularia tabacaria	1	92.54	622844.63
Acanthurus olivaceus	1	92.54	440293.88
Carcharhinus galapagensis	1	92.54	126513.74
Dasyatis centroura	1	92.54	52040.3

Species	Cluster	Occurrences	Landings
Paracaesio stonei	1	92.54	1.54
Solenidae	1	91.04	287220900.71
Asaphis violascens	1	91.04	295055.32
Cheimerius nufar	1	91.04	208179.26
Anchoa choerostoma	1	91.04	7735.71
Pontinus macrocephalus	1	91.04	6590.22
Plagioscion surinamensis	1	91.04	1254.54
Psettodes bennettii	1	91.04	1142.47
Anguilla	1	91.04	681.12
Harengula humeralis	1	91.04	289.74
Gymnothorax maderensis	1	91.04	0.04
Apsilus fuscus	1	89.55	26413299357.44
Plectropomus laevis	1	89.55	4489695.96
Heteropriacanthus cruentatus	1	89.55	1242758.78
Todaropsis eblanae	1	89.55	980573.49
Crenidens crenidens	1	89.55	784836.18
Serranus scriba	1	89.55	628832.49
Carcharhinus macloti	1	89.55	570429.77
Cookeolus japonicus	1	89.55	536480.25
Illex	1	89.55	518617.48
Sepia orbignyana	1	89.55	246948.91
Pomadasys rogerii	1	89.55	170482.16
Cepola	1	89.55	149720.06
Centrophorus	1	89.55	118223.95
Palinurus mauritanicus	1	89.55	116241.08
Iniistius pavo	1	89.55	115919.11
Cynoscion steindachneri	1	89.55	114556.35
Arnoglossus kessleri	1	89.55	3944.47
Arnoglossus thori	1	89.55	1401.34
Parupeneus pleurostigma	1	89.55	5.47
Palinurus	1	88.06	5638745.99
Epinephelus analogus	1	88.06	2497295.2
Pterygotrigla polyommata	1	88.06	890400.06
Sparisoma aurofrenatum	1	88.06	801069.08
Symphorichthys spilurus	1	88.06	0.06
Pomadasys argenteus	1	86.57	9216833.14
Scatophagus argus	1	86.57	361530.29
Patella	1	86.57	290528.65
Raja asterias	1	86.57	129018.48

Species	Cluster	Occurrences	Landings
Scorpis violacea	1	86.57	94786.07
Aetomylaeus bovinus	1	86.57	68264.22
Diplodus argenteus argenteus	1	86.57	29730.44
Acanthurus nigrofuscus	1	86.57	20029.48
Pristipomoides argyrogrammicus	1	86.57	1.03
Gomphosus varius	1	86.57	0.81
Gymnothorax rueppellii	1	86.57	0.72
Mycteroperca rubra	1	85.07	38642900.39
Lethrinus harak	1	85.07	11067260.37
Neocyttus rhomboidalis	1	85.07	5680354.52
Ephippidae	1	85.07	5130525.37
Dermatolepis dermatolepis	1	85.07	2239026.81
Psettodes belcheri	1	85.07	1816040.83
Sargocentron spiniferum	1	85.07	1597237.76
Acanthurus dussumieri	1	85.07	833626.82
Echeneidae	1	85.07	432647.15
Stellifer illecebrosus	1	85.07	345209.44
Moronidae	1	85.07	133828.17
Polydactylus approximans	1	85.07	112721.45
Aulostomidae	1	85.07	14.12
Melichthys vidua	1	85.07	0.52
Trochus niloticus	1	83.58	9580309.03
Seriola zonata	1	83.58	3918542.92
Campogramma glaycos	1	83.58	3019637.67
Triaenodon obesus	1	83.58	2252882.24
Cynoscion stolzmanni	1	83.58	1956969.59
Micropogonias altipinnis	1	83.58	1879473.59
Pempheridae	1	83.58	1676949.94
Alosa mediocris	1	83.58	1031709.36
Chaetodipterus zonatus	1	83.58	804274.43
Plectorhinchus schotaf	1	83.58	588538.94
Zebrasoma flavescens	1	83.58	18984.44
Haliporoides diomedeae	1	82.09	139757372.36
Alectis ciliaris	1	82.09	37756670.78
Pellonula leonensis	1	82.09	7449502.51
Alepes djedaba	1	82.09	6378631.91
Kyphosus cinerascens	1	82.09	4497031.33
Monotaxis grandoculis	1	82.09	2350684.44
Ostrea	1	82.09	1376703.46

Species	Cluster	Occurrences	Landings
Ophichthidae	1	82.09	1211086.26
Opsanus tau	1	82.09	482248.39
Nemipterus randalli	1	82.09	9154.2
Anchoa mitchilli	1	82.09	3399.33
Valamugil engeli	1	82.09	30.52
Scorpaenopsis diabolus	1	82.09	25.23
Sufflamen bursa	1	82.09	0.08
Chaetodon lunula	1	82.09	0.04
Cheilio inermis	1	82.09	0.02
Scyliorhinus stellaris	1	80.6	1835911.83
Lactophrys	1	80.6	1889.99
Paracaesio kusakarii	1	80.6	3.22
Arothron hispidus	1	80.6	0.65
Paracirrhites forsteri	1	80.6	0.44
Chaetodon unimaculatus	1	80.6	0.15
Novaculichthys taeniourus	1	80.6	0.03
Terapon jarbua	2	100	1362794392.9
Sphyrna lewini	2	100	270520207.28
Ostrea edulis	2	100	221105923.98
Hemiramphus brasiliensis	2	100	204243427.77
Alopias vulpinus	2	100	194368116.98
Acanthopagrus berda	2	100	168181952.26
Eucinostomus	2	100	163884598.19
Holocentridae	2	100	159653126.58
Lobotes surinamensis	2	100	119409701.98
Malacanthidae	2	100	94681045.25
Caranx ruber	2	100	79548797.04
Carcharhinus limbatus	2	100	75304609.53
Scophthalmus rhombus	2	100	73116926.21
Lutjanus gibbus	2	100	70045834.17
Epinephelus striatus	2	100	54745228.46
Polyprion americanus	2	100	51062077.96
Salmo trutta	2	100	48602336.19
Haemulon	2	100	47947805.3
Gnathanodon speciosus	2	100	45655290.54
Aprion virescens	2	100	42840448.42
Sparisoma viride	2	100	28799371.07
Carangoides orthogrammus	2	100	27785372.53
Trachinus draco	2	100	23048688.76

Species	Cluster	Occurrences	Landings
Salvelinus malma malma	2	100	21912559.49
Chaetodipterus faber	2	100	20647859.17
Centrolophidae	2	100	16963188.6
Palinurus elephas	2	100	16305991.25
Naso unicornis	2	100	13566112.4
Trachinus	2	100	13360120.42
Cephalopholis taeniops	2	100	8642476.16
Pseudupeneus maculatus	2	100	8051930.99
Mycteroperca xenarcha	2	100	7088643.2
Epinephelus polyphekadion	2	100	7021496.76
Carangoides malabaricus	2	100	6665523.37
Albulidae	2	100	6462975.85
Holocentrus adscensionis	2	100	5985724.49
Lutjanus fulvus	2	100	5920866.03
Isopisthus parvipinnis	2	100	5156927.85
Ctenochaetus striatus	2	100	4537187.32
Sciaena umbra	2	100	3667001.22
Lutjanus vivanus	2	100	3165221.08
Muraena helena	2	100	3122865.53
Leptoscarus vaigiensis	2	100	3080058.34
Epinephelus adscensionis	2	100	1846487.29
Diplodus puntazzo	2	100	1202058.7
Uranoscopus scaber	2	100	147191.85
Gymnura micrura	2	100	13446.15
Epinephelus marginatus	2	98.51	547030868.25
Homarus gammarus	2	98.51	243826815.07
Pristipomoides multidens	2	98.51	62285779.86
Peprilus paru	2	98.51	23819967.95
Scomberomorus regalis	2	98.51	17632779.67
Acanthurus lineatus	2	98.51	14071435.99
Plesionika edwardsii	2	98.51	13843649.82
Orthopristis chrysoptera	2	98.51	11472652.11
Chelon labrosus	2	98.51	9366164.03
Scorpaena porcus	2	98.51	8693240.6
Pomacanthus maculosus	2	98.51	3588874.7
Stereolepis gigas	2	98.51	2655603.13
Echidna nebulosa	2	98.51	1615360.12
Sepia latimanus	2	98.51	1615360.12
Epinephelus caninus	2	98.51	1133241.03

Species	Cluster	Occurrences	Landings
Centrophorus granulosus	2	98.51	532213.85
Panulirus polyphagus	2	97.01	151116130.99
Ostraciidae	2	97.01	50505965.41
Dicologlossa cuneata	2	97.01	47426252.45
Carcharhiniformes	2	97.01	35280792.92
Lutjanus erythropterus	2	97.01	14617401.65
Atherina presbyter	2	97.01	10032669.17
Terapontidae	2	97.01	9295667.91
Pomacentridae	2	97.01	5580778.21
Lophius budegassa	2	97.01	157586.41
Scyllaridae	2	95.52	1077046964.13
Rhinobatos	2	95.52	300045062.29
Palaemon serratus	2	95.52	200905018.7
Mustelus mustelus	2	95.52	72375194.02
Hyporthodus nigritus	2	95.52	16622970.51
Amphiarius phrygiatus	2	95.52	14149548.4
Hexanchus griseus	2	95.52	13948344.77
Scolopsis taeniata	2	95.52	8187890.33
Cephalopholis fulva	2	95.52	4524323.37
Negaprion brevirostris	2	95.52	4181615.93
Atractoscion aequidens	2	95.52	1454374.37
Scyllarides latus	2	95.52	118481.76
Centropomus undecimalis	2	94.03	933898100.52
Panulirus longipes	2	94.03	592336962.03
Gempylidae	2	94.03	318094139.26
Lichia amia	2	94.03	186130413.35
Larimus breviceps	2	94.03	102154840.56
Xiphiidae	2	94.03	85267013.09
Epinephelus itajara	2	94.03	74330913.89
Lota lota	2	94.03	16852769.66
Etelis oculatus	2	94.03	9668906.82
Somniosus microcephalus	2	94.03	6261031.94
Haliotis	2	92.54	438252857.9
Crassostrea rhizophorae	2	92.54	345242189.72
Sparidentex hasta	2	92.54	167590905.05
Scarus	2	92.54	93554452.86
Sander lucioperca	2	92.54	80549221.08
Lethrinus olivaceus	2	92.54	79785362.49
Belonidae	2	92.54	26439403.12

Species	Cluster	Occurrences	Landings
Dipturus batis	2	92.54	15527000.26
Saxidomus gigantea	2	92.54	11460859.13
Tridacna maxima	2	92.54	8049210.79
Trigla lyra	2	92.54	2591840.67
Rachycentridae	2	91.04	207246030.72
Rhabdosargus sarba	2	91.04	147740963.17
Kajikia albida	2	91.04	101616409.13
Harengula clupeola	2	91.04	96173662.9
Rhizoprionodon acutus	2	91.04	89181593.8
Pristipomoides filamentosus	2	91.04	80429729.18
Lagocephalus laevigatus	2	91.04	58513328.51
Scorpaena scrofa	2	91.04	8207306.95
Hyporthodus niveatus	2	91.04	5724777.48
Donax trunculus	2	91.04	4912953.81
Leucoraja fullonica	2	91.04	1120504.75
Farfantepenaeus brasiliensis	2	89.55	887598002.52
Kyphosidae	2	89.55	596046674.29
Latidae	2	89.55	154457115.01
Drepane punctata	2	89.55	83616805.15
Istiophorus	2	89.55	59626344.95
Pseudocaranx dentex	2	89.55	48138407.46
Lampris guttatus	2	89.55	30530654.18
Diodon holocanthus	2	89.55	1193201.97
Panulirus	2	88.06	924454402.04
Scomberomorus sierra	2	88.06	305969947.27
Calamus	2	88.06	257543550.63
Osmerus mordax mordax	2	88.06	171719019.28
Ensis siliqua	2	88.06	109554391.17
Anguilla rostrata	2	88.06	70316314.44
Panulirus homarus	2	88.06	59166155.21
Valamugil	2	88.06	9831263.5
Hyporhamphus	2	88.06	9806492.31
Sphyrna tiburo	2	88.06	3091863.06
Scorpaenichthys marmoratus	2	88.06	2125653.48
Sphyraena guachancho	2	88.06	8028.39
Myliobatis aquila	2	88.06	2499.12
Megalops atlanticus	2	86.57	537347174.7
Mycteroperca bonaci	2	86.57	200340825.23
Mycteroperca	2	86.57	130119174.36

Species	Cluster	Occurrences	Landings
Hyperoglyphe antarctica	2	86.57	85782165.68
Scomberesox saurus saurus	2	86.57	78994581.33
Squatinidae	2	86.57	71575421.93
Mulloidichthys martinicus	2	86.57	7816035.14
Haemulon bonariense	2	86.57	4389884.37
Semicossyphus pulcher	2	86.57	1473696.92
Acanthuridae	2	85.07	1347865983.05
Farfantepenaeus brevirostris	2	85.07	426607455.35
Dicentrarchus	2	85.07	112352243.21
Carcharhinus plumbeus	2	85.07	40340163.16
Latridae	2	85.07	39258038.29
Lethrinus microdon	2	85.07	2174862.29
Ctenosciaena gracilicirrhus	2	85.07	64801.49
Stellifer rastrifer	2	85.07	2708.3
Donax	2	83.58	389035827.99
Tridacna	2	83.58	85718811.34
Triakidae	2	83.58	78758093.52
Trigla	2	83.58	38834199.1
Etelis	2	83.58	11767339.29
Hyporthodus flavolimbatus	2	83.58	6917090.89
Lutjanus peru	2	82.09	863328642.27
Pagrus caeruleostictus	2	82.09	626627441.57
Atherinidae	2	82.09	550501749.08
Acanthurus	2	82.09	260358338.43
Mustelus	2	82.09	226013529.36
Necora puber	2	82.09	143324853.65
Girella tricuspidata	2	82.09	53902708.87
Cyprinus carpio	2	82.09	28578944.02
Cynoscion reticulatus	2	82.09	17146184.74
Scarus ghobban	2	82.09	8717170.12
Panulirus penicillatus	2	82.09	3380935.09
Diodon	2	82.09	427539.41
Murex	2	80.6	652920062.12
Solea	2	80.6	514736306.97
Istiophorus albicans	2	80.6	338294104.71
Sphyrna	2	80.6	316945203.51
Pomadasys incisus	2	80.6	93711682.44
Chelon auratus	2	80.6	54215043.76
Sphyraena obtusata	2	80.6	49852396.22

Species	Cluster	Occurrences	Landings
Gymnothorax	2	80.6	5932818.73
Haemulon aurolineatum	2	80.6	4978752.21
Soleidae	3	100	60662801883.43
Caranx hippos	3	100	14060861852.64
Sparus aurata	3	100	4386741193.38
Alopias	3	100	4076729794.71
Lithognathus mormyrus	3	100	2897770062.39
Microstomus kitt	3	100	2436734573.72
Diplodus sargus sargus	3	100	1220362998
Perca fluviatilis	3	100	984738792.6
Istiophoridae	3	100	983582389.19
Dentex gibbosus	3	100	741041950.65
Bothidae	3	100	711379612.63
Esox lucius	3	100	647305986.2
Sillago sihama	3	100	588081437.04
Decapterus macarellus	3	100	511576981.81
Cynoscion virescens	3	100	331321622.39
Syacium ovale	3	100	173179904.94
Epinephelus malabaricus	3	100	104615659.28
Lamnidae	3	100	96874125.97
Diodontidae	3	100	87876722.97
Acipenser oxyrinchus	3	100	45791343.25
Serranus cabrilla	3	100	45147919.74
Carcharhinus longimanus	3	100	15093092.03
Torpedinidae	3	100	11377065.33
Pagrus africanus	3	100	804283.97
Bolinus brandaris	3	100	553982.16
Pseudorhombus arsius	3	100	55.47
Gastropoda	3	98.51	19468659149.9
Glyptocephalus cynoglossus	3	98.51	10295819301.63
Platichthys flesus	3	98.51	8535963718.52
Mustelus schmitti	3	98.51	3272735546.14
Pollachius pollachius	3	98.51	2379927440.68
Platycephalus indicus	3	98.51	1586720662.02
Genyatremus luteus	3	98.51	316150295.8
Squaliformes	3	98.51	146179159.24
Nephropidae	3	98.51	34691237.73
Sphoeroides	3	98.51	97380.52
Carcharhinus falciformis	3	97.01	3905955365.34

Species	Cluster	Occurrences	Landings
Acanthocybium solandri	3	97.01	2976292682.58
Sillaginidae	3	97.01	2760401332.65
Scaridae	3	97.01	2484586022.87
Diplodus vulgaris	3	97.01	2460892807.43
Trachurus picturatus	3	97.01	851507113.83
Paralabrax maculatofasciatus	3	97.01	216148996.97
Balistes polylepis	3	97.01	203995133.1
Porichthys analis	3	97.01	177186252.16
Achirus mazatlanus	3	97.01	93040460.85
Muraena augusti	3	97.01	16950223.84
Euthynnus alletteratus	3	95.52	7406474556.53
Mullus	3	95.52	5247311995.22
Limanda limanda	3	95.52	3700935086.08
Peprilus	3	95.52	322759285.49
Stephanolepis hispidus	3	95.52	4237318.75
Mullus surmuletus	3	94.03	6479506350.77
Chloroscombrus chrysurus	3	94.03	3903010993.25
Phycis phycis	3	94.03	123767874.83
Coris julis	3	94.03	12311433.27
Plotosus lineatus	3	94.03	11833036.08
Tetraodontidae	3	92.54	17753755537.1
Sphyraena barracuda	3	92.54	5781328773.98
Rhinobatidae	3	92.54	2001731368.7
Trachurus lathami	3	92.54	1743452880.66
Oblada melanura	3	92.54	667194304.84
Menticirrhus americanus	3	92.54	612113738.57
Drepane africana	3	92.54	593685394.84
Citharus linguatula	3	92.54	47659315.79
Parupeneus barberinus	3	92.54	733143.94
Triglidae	3	91.04	29692575245.04
Auxis rochei	3	91.04	16274720025.71
Argentina	3	91.04	3159083970.69
Synodus scituliceps	3	91.04	511079543.63
Scyliorhinus canicula	3	91.04	463364798.45
Octopus cyanea	3	91.04	426706280.95
Congresox talabonoides	3	91.04	425466036.42
Macroramphosus scolopax	3	91.04	395329860.33
Trachinidae	3	91.04	91465627.35
Naucrates ductor	3	91.04	48045322.49

Species	Cluster	Occurrences	Landings
Prionotus	3	91.04	17316535.09
Squalus	3	91.04	1156430.37
Gymnothorax unicolor	3	91.04	18628.61
Sepia officinalis	3	89.55	6470866291.61
Dicentrarchus labrax	3	89.55	4070093574.16
Litopenaeus schmitti	3	89.55	2538136916.18
Scorpaena sonorae	3	89.55	520412589.22
Aspistor quadriscutis	3	89.55	369695272.85
Muraenidae	3	89.55	272880679.34
Zeidae	3	89.55	10483941.54
Maja squinado	3	89.55	3867241.61
Mobula mobular	3	89.55	2959290.61
Panulirus argus	3	88.06	14778066750.77
Sphyraenidae	3	88.06	6075000682.41
Lepidorhombus whiffiagonis	3	88.06	2009840203.26
Dasyatidae	3	88.06	1928316233.82
Dentex dentex	3	88.06	1762277496.63
Centropomus	3	88.06	1255787047.39
Apostichopus japonicus	3	88.06	869730176.87
Eopsetta jordani	3	88.06	262317083.5
Lethrinus miniatus	3	88.06	30891747.09
Rostroraja alba	3	88.06	453235.69
Octopus	3	86.57	69903155600.08
Rajiformes	3	86.57	60406186400.95
Lobatus gigas	3	86.57	11188647541.33
Myliobatidae	3	86.57	3235893035.85
Pseudotolithus senegalensis	3	86.57	1909003636.82
Pomadasys kaakan	3	86.57	924624408.74
Paralichthys	3	86.57	371697750.84
Scylla	3	86.57	98949905.74
Cynoglossidae	3	86.57	48918026.19
Menticirrhus saxatilis	3	86.57	3545157.45
Thalassoma pavo	3	86.57	1280058.7
Chromis chromis	3	86.57	1157
Haemulidae	3	85.07	74245725015.9
Anarhichas lupus	3	85.07	33300654250.8
Diplodus	3	85.07	20398841317.57
Lactarius lactarius	3	85.07	11498882507.33
Conger conger	3	85.07	9027298530.71

Species	Cluster	Occurrences	Landings
Scorpaenidae	3	85.07	3796715491.61
Lutjanus purpureus	3	85.07	3314798865.77
Haliotidae	3	85.07	1618755146.27
Orthopristis reddingi	3	85.07	720396346.36
Eledone	3	85.07	533260316.88
Parophrys vetulus	3	85.07	390848524.93
Salvelinus alpinus alpinus	3	85.07	203265704.04
Portunus armatus	3	85.07	88396966.34
Argentina sphyraena	3	85.07	36524213.48
Sphoeroides marmoratus	3	85.07	512634.58
Pontinus kuhlii	3	85.07	169236.59
Pelates quadrilineatus	3	85.07	14932.17
Lepidorhombus boscii	3	85.07	13256.96
Molva dypterygia	3	83.58	3291275339.8
Scomber australasicus	3	83.58	2618574355.93
Diplodus annularis	3	83.58	472607204.45
Acanthocardia tuberculata	3	83.58	203131864.35
Gadiformes	3	82.09	9058518644.49
Micropogonias	3	82.09	3019414972.58
Serranus	3	82.09	1157324487.11
Ophidiidae	3	82.09	91126982.29
Eucinostomus argenteus	3	82.09	294637.16
Cepola macrophthalma	3	82.09	35741.4
Pagellus erythrinus	3	80.6	6101606289.13
Genypterus capensis	3	80.6	2206223933.97
Epinephelus morio	3	80.6	2017193302.58
Metapenaeus affinis	3	80.6	1545466715.37
Scomberomorus tritor	3	80.6	1494142011.77
Chelidonichthys cuculus	3	80.6	1483888485.07
Coregonus	3	80.6	1022358784.86
Squalus suckleyi	3	80.6	831634733.43
Dorosoma	3	80.6	5613.18
Engraulis ringens	4	100	2877198119530179
Theragra chalcogramma	4	100	510868614798430
Sardinops sagax	4	100	352686648969777
Marine fishes not identified	4	100	144390862409846
Gadus morhua	4	100	48979734887457.3
Clupea harengus	4	100	42015905842750.1
Scyphozoa	4	100	34074191691020.9

Species	Cluster	Occurrences	Landings
Scomber	4	100	33992825125336.5
Nemipteridae	4	100	30642267967139.7
Rastrelliger kanagurta	4	100	23215233848100.4
Mollusca	4	100	23110162123049.4
Brevoortia patronus	4	100	22402770987341.3
Sardinella	4	100	16262304861048.3
Trichiurus lepturus	4	100	16154577290815.4
Sardina pilchardus	4	100	16031679013584.5
Marine finfishes not identified	4	100	15419270370067.1
Engraulis japonicus	4	100	15291420570012
Engraulidae	4	100	14335452707640.1
Micromesistius poutassou	4	100	13797155857988.3
Katsuwonus pelamis	4	100	12487524715538
Merluccius hubbsi	4	100	10398494127424.1
Todarodes pacificus	4	100	10099162290182.3
Clupeidae	4	100	9163715287380.66
Scomber japonicus	4	100	8684562107637.69
Carangidae	4	100	7979083375497.08
Acetes japonicus	4	100	7651856582445.44
Scomber scombrus	4	100	7527391077754.71
Clupea pallasii pallasii	4	100	7527073433165.71
Crassostrea virginica	4	100	6480519622980.13
Miscellaneous marine crustaceans	4	100	5862755005640.29
Cololabis saira	4	100	5668208059753.45
Pleuronectidae	4	100	5324499873012.34
Engraulis encrasicolus	4	100	5254303917244.63
Oncorhynchus gorbuscha	4	100	5090520215964.52
Trachurus japonicus	4	100	4842043515312.04
Thunnus albacares	4	100	4590273220712.96
Trachurus capensis	4	100	4449121190396.14
Scomberomorus	4	100	4026350355838.91
Merluccius	4	100	3767912035635.13
Sprattus sprattus	4	100	3642303488155.13
Penaeidae	4	100	3522192726202.26
Decapterus	4	100	3458365898840.78
Synodontidae	4	100	3390803172237.35
Gadus macrocephalus	4	100	3344628420438.27
Sardinella longiceps	4	100	3294553531129.12
Sciaenidae	4	100	3209517883951.13

Species	Cluster	Occurrences	Landings
Melanogrammus aeglefinus	4	100	3156309235538.06
Pollachius virens	4	100	3053175041248.38
Miscellaneous aquatic invertebrates	4	100	2937907076082.57
Decapoda	4	100	2597718133804.6
Bivalvia	4	100	2433630903885.34
Oncorhynchus	4	100	2379825912653.68
Nemipterus	4	100	2251516691140.62
Decapterus macrosoma	4	100	1946277008191.53
Rastrelliger brachysoma	4	100	1809130975646.22
Rastrelliger	4	100	1769188073455.09
Sardinella aurita	4	100	1660920859991.95
Stolephorus	4	100	1502483286762.16
Placopecten magellanicus	4	100	1484105776429.83
Trachurus	4	100	1390583843896.52
Thunnus obesus	4	100	1388193752925.41
Scombridae	4	100	1364905886157.32
Spisula solidissima	4	100	1222605147344.73
Pleurogrammus azonus	4	100	1193613725094.47
Pleuronectiformes	4	100	1106196033864.82
Clupeiformes	4	100	1102413191852
Portunus pelagicus	4	100	1057898103610.55
Merluccius merluccius	4	100	1021120955472.46
Larimichthys polyactis	4	100	998733689341.03
Harpadon nehereus	4	100	984412064625.74
Trachurus trecae	4	100	965570848544.65
Trachurus trachurus	4	100	958801497259.44
Selaroides leptolepis	4	100	943552781353.57
Sardinella brasiliensis	4	100	887738203549.1
Sepiida	4	100	870724685211.56
Ethmalosa fimbriata	4	100	870347598069.95
Oncorhynchus keta	4	100	784999520577.06
Sparidae	4	100	762825752941.55
Octopoda	4	100	680787683189.41
Oncorhynchus nerka	4	100	655470655458.43
Ammodytes personatus	4	100	653501406393.43
Larimichthys crocea	4	100	629056734927.7
Thunnus alalunga	4	100	626874166418.61
Sardinella lemuru	4	100	591594041397.77
Laemonema longipes	4	100	561348816763.54

Species	Cluster	Occurrences	Landings
Scomberomorus commerson	4	100	502775162808.67
Callinectes sapidus	4	100	492184221111.41
Elasmobranchii	4	100	452928433945.65
Platycephalidae	4	100	449458183935.02
Penaeus	4	100	432763135532.38
Mytilus edulis	4	100	400085259248.07
Sardinella maderensis	4	100	394408463249.54
Megalaspis cordyla	4	100	390517729968.43
Pleuronectes platessa	4	100	382242352137.2
Merluccius bilinearis	4	100	334185346072.1
Chionoecetes opilio	4	100	273685610486.52
Mugil cephalus	4	100	243563918974.55
Merluccius gayi gayi	4	100	233986167337.23
Homarus americanus	4	100	230495432047.46
Paralithodes camtschaticus	4	100	223691875042.34
Trachurus mediterraneus	4	100	209589637397.45
Siluriformes	4	100	209537076590.17
Ariidae	4	100	203363077373.92
Sarda sarda	4	100	191258154851.1
Mercenaria mercenaria	4	100	166751716525.33
Acetes	4	100	145961451781.08
Micropogonias furnieri	4	100	143550278081.17
Sebastes norvegicus	4	100	141686180990.78
Sardinella fimbriata	4	100	131542703865.42
Pomatomus saltatrix	4	100	123663507988.69
Pennahia argentata	4	100	113136433713.94
Eleginus gracilis	4	100	95715659980.52
Fenneropenaeus indicus	4	100	78554549196.05
Hippoglossus stenolepis	4	100	73297376882.82
Pseudopleuronectes herzensteini	4	100	62148883258.12
Oncorhynchus kisutch	4	100	51184106469.48
Boops boops	4	100	43657995584.86
Molva molva	4	100	42021790451.06
Konosirus punctatus	4	100	40854425687.35
Thyrsites atun	4	100	37323638668.46
Litopenaeus stylirostris	4	100	35863993862.91
Pagrus major	4	100	35090865736.58
Exocoetidae	4	100	32358904112.51
Metacarcinus magister	4	100	30259200043.73

Species	Cluster	Occurrences	Landings
Oncorhynchus tshawytscha	4	100	25450906126.33
Farfantepenaeus californiensis	4	100	22168012587.19
Farfantepenaeus aztecus	4	100	21997016699.65
Fistularia corneta	4	100	17914653092.29
Paralichthys dentatus	4	100	16571106806.57
Scomberomorus maculatus	4	100	15065134216.26
Turbo cornutus	4	100	9350721140.77
Cynoscion nebulosus	4	100	8600504286.29
Panulirus cygnus	4	100	7100011846.37
Squilla mantis	4	100	5797104922.67
Sarotherodon galilaeus	4	100	2783126026.05
Lepturacanthus savala	4	100	2745119771.84
Polynemus paradiseus	4	100	362639122.02
Mallotus villosus	4	98.51	60124460417897.3
Leiognathidae	4	98.51	17351136392848.1
Brevoortia tyrannus	4	98.51	8425000982655.33
Pleurogrammus monopterygius	4	98.51	235084481493.97
Cephalopholis boenak	4	98.51	44332490663.29
Nemipterus virgatus	4	98.51	40332967900.87
Aulacomya ater	4	98.51	27926959103.27
Selene peruviana	4	98.51	26026850461.51
Bathylagidae	4	98.51	5935364128.24
Lateolabrax japonicus	4	98.51	5283400787.27
Portunus trituberculatus	4	97.01	3255221360900.82
Sebastes	4	97.01	2875453531044.98
Trisopterus esmarkii	4	97.01	2381287726184.64
Trachysalambria curvirostris	4	97.01	2057600033201.3
Octopus vulgaris	4	97.01	123792222233.28
Saurida tumbil	4	97.01	118962425499.72
Scomberomorus niphonius	4	97.01	104767238146.34
Brosme brosme	4	97.01	37592436384.44
Thunnus orientalis	4	97.01	20318468926.29
Tenualosa ilisha	4	95.52	1359323431727.81
Thunnus maccoyii	4	95.52	124344198882.75
Engraulis anchoita	4	95.52	40198878361.83
Xiphopenaeus kroyeri	4	95.52	28364659542.71
Microstomus pacificus	4	95.52	8491774848.61
Trachurus murphyi	4	94.03	84253314469716.3
Muraenesox cinereus	4	94.03	1736037341520.6

Species	Cluster	Occurrences	Landings
Netuma thalassina	4	94.03	241408241981.3
Arctoscopus japonicus	4	94.03	34111885344.84
Chirocentrus	4	94.03	16169473235.64
Priacanthus	4	92.54	652822557561.78
Merlangius merlangus	4	92.54	361395152113.66
Leiostomus xanthurus	4	92.54	3079857079.88
Marine pelagic fishes not identified	4	91.04	5105987323410.28
Sepiidae	4	91.04	728573357220.25
Coryphaena hippurus	4	91.04	138834590890.83
Carcharhinidae	4	91.04	87608173618.85
Lutjanus malabaricus	4	91.04	25065158906.79
Odontamblyopus rubicundus	4	91.04	17495546637.09
Pterotolithus maculatus	4	91.04	13166839690.39
Litopenaeus vannamei	4	91.04	12727998088.16
Setipinna taty	4	91.04	5753132064.59
Coilia dussumieri	4	91.04	1997809163.46
Cynoglossus cynoglossus	4	91.04	1421162537.99
Escualosa thoracata	4	91.04	773670047.37
Parapristipoma trilineatum	4	91.04	615940000
Metapenaeus brevicornis	4	91.04	364910568.05
Secutor insidiator	4	91.04	209822237
Parapenaeopsis uncta	4	91.04	174823890.82
Doryteuthis opalescens	4	89.55	155435358712.35
Scomber colias	4	89.55	134958286014.27
Teuthida	4	88.06	3074615789654.76
Pectinidae	4	88.06	2378553930213.21
Apogonidae	4	88.06	55966795944.58
Micropogonias undulatus	4	88.06	11956873569.95
Osteichthyes	4	88.06	7652907753.31
Spicara smaris	4	88.06	7210279822.43
Auxis	4	86.57	609975162058.59
Pseudotolithus	4	86.57	307497781322.91
Ethmidium maculatum	4	86.57	21368525435.41
Psenopsis anomala	4	86.57	16579284931.16
Lepidorhombus	4	86.57	6217462810.24
Amblygaster sirm	4	86.57	2253142531.71
Dendrobranchiata	4	85.07	843540859040.9
Sarda chiliensis	4	85.07	233217215933.85
Reinhardtius hippoglossoides	4	85.07	158110437042.21

Species	Cluster	Occurrences	Landings
Scomberomorus guttatus	4	85.07	149876400231.1
Chondrichthyes	4	85.07	145643996476.98
Thunnus tonggol	4	85.07	51160241728.47
Parapenaeopsis	4	85.07	47843410416.47
Cetengraulis mysticetus	4	83.58	579740735128.1
Engraulis mordax	4	83.58	455569854772.27
Conger myriaster	4	83.58	18466662785.99
Paralichthys olivaceus	4	83.58	8029858012.69
Scomberomorus brasiliensis	4	83.58	7214971933.91
Eledone cirrhosa	4	83.58	1111804999
Hexagrammos otakii	4	83.58	317040000
Aristaeomorpha foliacea	4	83.58	205272381.07
Mya arenaria	4	83.58	101274911.48
Fenneropenaeus chinensis	4	82.09	266776052074.28
Seriola lalandi	4	82.09	147077574025.74
Clupanodon thrissa	4	82.09	57693032821.29
Xiphias gladius	4	82.09	50782246896.25
Batoidea	4	82.09	41096236289.51
Pseudopleuronectes americanus	4	82.09	14574832214.62
Todarodes sagittatus	4	82.09	5419440561.72
Dentex tumifrons	4	82.09	210530000
Engraulis capensis	4	80.6	11274781096322.4
Mytilus galloprovincialis	4	80.6	151577361620.62
Dussumieria	4	80.6	71198606881.98
Anoplopoma fimbria	4	80.6	55153369924.1
Anadara	4	80.6	42019440118.69
Charybdis hellerii	4	80.6	0

Chapter 5

Reducing the environmental impact of food consumption through fiscal policies: the case of Spain[1]

The environmental footprint generated through human food consumption intensifies climate change and food insecurity. This analysis searches for the optimal fiscal policies minimizing the environmental impact (Carbon Footprint, Water Footprint and Food Loss and Waste) of food consumption by promoting less harmful patterns. This analysis consists on a two-step procedure. Firstly, the effect of fiscal policies on food consumption is estimated through Almost Ideal Demand Systems. Secondly, the optimal fiscal policies minimizing the environmental impact are obtained solving an optimization problem. The data of this analysis corresponds to the monthly Spanish food consumption from 2005 to 2021. Main findings from this analysis reflect the effectiveness of fiscal policies to decrease the environmental impact, especially when the target is unidimensional and food consumption is completely represented. In particular, fiscal policies between -20 and 20% of the food prices may reduce environmental impact between 9 and 18% depending on the dimensions targeted.

5.1 Introduction

The rapid evolution of climate change has raised concerns on the environmental and human consequences derived from this global process. International initiatives and agreements have been signed for the development of effective strategies targeting the mitigation of the climate variations as well as its effects (United Nations, 2015; UN, 2015). Even though greenhouse gas emissions play the major role in the objectives set within these international agreements, additional environmental aspects such as the use of natural resources like water should not be disregarded in the evaluation of the environmental impact derived from human activities (Fang et al., 2014; Page et al., 2012).

The environmental footprints contribute to measure the environmental impact associated to human activities in a particular environmental dimension (Hoekstra and Wiedmann, 2014). Rather than looking at one indicator in a particular period of time, footprints allow to combine several environmental outcomes arising in the different stages of the human activity, providing measures with more complete information (International Organization for Standardization, 2018; Hoekstra et al., 2011). Environmental footprints are widely used by public institutions and previous literature as instruments to quantify the impact of human activities in specific environmental

[1]This analysis has been presented in the *X AERNA Conference (2022)*.

dimensions (MITECO, 2021; EPA, 2022; BOE, 2014; Carballo-Penela et al., 2012; Yu et al., 2010). Regarding greenhouse gas emissions, the Carbon Footprint (CF) is used to assess their CO_2 equivalent emissions generated in the process of the activity. In the case of water, the Water Footprint (WF) measures the surface, underground and rain water required for the activity in addition to the water needed to assimilate the pollution generated (Hoekstra et al., 2011).

The environmental consequences of the human food consumption in terms of greenhouse gas emissions and water use have been documented in previous literature (Aguilera et al., 2021; Esteve-Llorens et al., 2021; Blas et al., 2019; Vanham et al., 2013; Mekonnen and Hoekstra, 2012, 2011). Nevertheless, these effects could be larger if the Food Loss and Waste (FLW) generated along the production of the food consumed is considered. FLW are estimated to be around one third of the human food consumption (Gustavsson et al., 2011). These lost resources not only threaten food security by reducing the available food, but are also responsible for the environmental impact caused in their production (Cattaneo et al., 2021). The studies on food ecolabeling reflect that people are concerned with the environmental implications of their food consumption (Muller et al., 2019) accepting price variations to turn it more sustainable in some cases (Onofri et al., 2018). In particular, consumers are found to be aware of the CF (Rondoni and Grasso, 2021; Canavari and Coderoni, 2020; Liu et al., 2017), WF (Naseer et al., 2020; Pomarici et al., 2018) and FLW (Obuobi et al., 2022; Gracia and Gómez, 2020) derived from their food consumption and show some predisposition to decrease these environmental impacts.

The agricultural sector is globally protected by border and market policies, fiscal subsidies to production and demand and the support of general services. Most of the support provided to this sector is oriented to producers and specific groups of food such as rice, sugar and meat. Reorienting the subsidies of producers to consumers would have larger positive implications for the consumption of healthy food although it could have negative environmental and economic consequences. Therefore, enlarging the price incentives derived from the control of border and market prices seems more adequate to modify consumption patterns as carbon emissions could be reduced as well avoiding to face the economic trade-offs arisen with direct subsidies (FAO et al., 2022). Indeed, taxes elevating the price of different animal-sourced food products have been found to decrease greenhouse gas emissions by discouraging their consumption (Forero-Cantor et al., 2020; Säll and Gren, 2015). The present analysis aims to expand previous literature by considering several environmental dimensions in addition to the CF such as the WF, the FLW and a combination of these three dimensions. In addition, all major food categories (Gustavsson et al., 2011) are considered in order to reflect more accurately the food consumption and increase the efficacy of the fiscal policies by influencing simultaneously on multiple harmful categories while providing more environmentally friendlier food alternatives. Furthermore, the present analysis searches the optimal combination of taxes and subsidies that encourages consumers to modify consumption towards food diets minimizing the environmental dimensions studied rather than evaluating the impact of specific fiscal policies.

The present analysis focuses on the case of Spain. This country has experienced an increase in the intensity and frequency of heatwaves (responsible of more than 1,800 deaths from 2015 to 2020) and the occurrences of intense precipitations during long periods of time as a result of the climate change (AEMET, 2021). However, further consequences are expected to arise with the evolution of climate change in Spain like a significant reduction in the availability of water resources, changes in the distribution of species and landscapes, losses in biodiversity and human health harms (MITECO, 2020). In order to prevent the devastating consequences of these events and comply with the international environmental agreements joined, the Spanish National Climate

Change Adaptation Plan 2021 - 2030 (NCCAP 2021-2030) has been elaborated (MITECO, 2020). This plan aims to improve the understanding and measurement of the climate change causes, their processes and their consequences as well as to define strategies for their mitigation. As a Member of the European Union, most pollutant activities of Spain are regulated by the European Trading System (ETS) (EC, 2003). The ETS establishes emission limits for each firm and allows to exchange them in case of excess or lack. However, the total emission allowed are gradually decreased to reduce the greenhouse gas emissions produced in the region. In addition, each Member State must reduce by 2030 the emissions of activities non-regulated by the ETS according to the quantity established. In the case of Spain, 2005 greenhouse gas emissions must be reduced in 26% (EC, 2018).

A two-step methodology is applied to obtain the optimal fiscal polices minimizing the environmental dimensions studied. Firstly, since fiscal policies are instrumented through food price variations, their effect on consumption is estimated using Almost Ideal Demand Systems (AIDS) (Deaton and Muellbauer, 1980a). Secondly, based on the environmental impact of food consumption and the consumption resulting from the implementation of fiscal policies, the set of taxes and subsidies minimizing the environmental dimension chosen is obtained. Even though each environmental dimension requires from different fiscal policies to be minimized, the optimal combinations of taxes and subsidies seem determined by the average environmental impact of the corresponding food category and how its consumption is affected by variations in the food prices. When fiscal policies are constraint to a social acceptable limit of -20 and 20%, the reduction of the environmental impact achieved ranges between 9 and 18% depending on the preferences for the environmental dimensions analyzed. Robustness checks are additionally performed considering lower limits for the fiscal policies and their implementation through single food categories.

The present article is structured as follows. Section 5.2 presents the data on consumption and environmental impact used for the analysis. Section 5.3 describes the methodology applied in the analysis. Section 5.4 shows the results obtained in each step of the analysis. Section 5.5 discusses the findings of this analysis and compares them with those from the previous literature. Section 5.6 summarizes the main findings of the approach followed in this analysis.

5.2 Environmental consequences of food consumption in Spain

This section focuses on how food consumption in Spain has contributed to environmental damages in the two last decades. After describing the data used in the analysis (Subsection 5.2.1), issues such as the food basket composition during this period (Subsection 5.2.2) and the environmental consequences of the food consumed in terms of footprints (Subsection 5.2.3) are addressed.

5.2.1 Data on food consumption

Information on the food consumption of the Spanish households has been extracted from the Panel of Household Food Consumption, elaborated by the Spanish Ministry of Agriculture, Fishing and Food (MAPAMA, 2021). This data set reports detailed information on the monthly food consumption of households, specifying the quantity of food consumed as well as its price and consequent expenditure for more than 460 food categories. The period of time considered in this analysis starts in January 2005 and ends in November 2021.

Consumption in the sampled period is measured in kilograms and liters depending on the type of food. In order to homogenize the unit of measurement for further analysis, the quantity and prices of liquid food have been converted to kg and Euros/kilogram using the volume conversion factor provided by Charrondiere et al. (2011).

Since working with the large amount of food categories specified in the Panel of Household Food Consumption may harden the implementation of policies as well as overshadow the main findings of this analysis, food categories have been simplified using the commodity classification proposed in Gustavsson et al. (2011) following the major categories of the Food Balance Sheets (FAOSTAT, 2022): *cereals*, *fish & seafood*, *fruit & vegetables*, *meat*, *dairy & eggs*, *oilseeds & pulses*, *roots & tubers* and *remaining food* (including products as pastries, oil, beverages, sugar, coffee and chocolate, among others). See Appendix 5.A for a detailed classification of the categories in MAPAMA (2021). The prices for the eight major categories have been calculated as the weighted average of the prices from their subcategories.

5.2.2 Description of the Spanish food consumption

According to FAO et al. (2022), Spain resembles the high-income countries profile regarding nourishment, food insecurity and extreme weights. The prevalence of undernourishment both, in Spain and the high-income countries, is lower than 2.5%, below the 9% global prevalence. Food insecurity is slightly above the corresponding to high-income countries in severe (2.0% prevalence) and moderate levels (8.6% prevalence). Although Spain's prevalence of obesity (23.8%) almost doubles the global figure (13.1%), it remains below the high income countries (24.3%). By contrast, the prevalence low weight at birth (8.3%) is higher than high-income countries (7.6%) and lower than the global one (14.6%). In particular, the average monthly consumption of a person in Spain in 2021 is about 53 kilograms of food, having decreased in around 1.5 kilograms with respect to the average monthly consumption in 2005.

Table 5.1: Composition of the average monthly Spanish food basket.

	2005		2021		Benchmark basket	
Food category	Kg	(%)	Kg	(%)	Kg	(%)
Cereals	4.79	8.76	3.63	6.79	4.12	7.53
Dairy & eggs	11.05	20.18	9.78	18.31	10.14	18.57
Fish & seafood	2.44	4.46	1.92	3.60	2.27	4.15
Fruit & vegetables	13.68	24.98	13.90	26.03	14.41	26.38
Meat	4.45	8.12	3.81	7.14	4.31	7.89
Oilseeds & pulses	0.74	1.36	0.84	1.57	0.75	1.38
Roots & tubers	2.60	4.75	2.44	4.57	2.53	4.63
Remaining food	15.00	27.40	17.07	31.98	16.10	29.47

Table 5.1 presents the composition, in kilograms and percentage, of the average monthly food basket of an Spanish person in 2005, 2021 and the period between these years. The latter is denoted as *benchmark basket* since it represents the average monthly food basket of an Spanish person during the period analyzed. The relative composition of the average monthly food basket seems to be stable over the period analyzed. *Fruit & vegetables* as well as *dairy & eggs* are amongst the categories most consumed, representing over 26 and 18% of

the *benchmark basket*, respectively. *Meat* and *cereals* are the following categories most consumed, representing over 7% of this basket each. The contribution of *fish & seafood* and *roots & tubers* is slightly lower, representing around 4% each. The category least consumed is *oilseeds & pulses*, with a contribution of around 1%. The conglomerate of products in the *remaining food* category constituted the one third left of the *benchmark basket*. Nevertheless, significant variations are observed from 2005 to 2021 in the quantity of food consumed for some categories. In general, consumption has decreased in almost all categories, specially in *cereals* and *dairy & eggs* (it has decreased in more than 1 kilogram in each category). The only exceptions for which consumption has increased are *fruit & vegetables*, *oilseeds & pulses* and *remaining food*, having the latter the largest positive variation (over 2 kilograms).

5.2.3 Environmental consequences

The consumption of food causes multiple environmental side-effects along the entire food chain, from the cultivation of raw products to the final household consumption. Since studying the consequences on several environmental dimensions allows to provide more precise estimates of the impact produced by food consumption (Fang et al., 2014), the present analysis considers the tonnes of equivalent CO_2 emissions (Carbon Footprint, CF), the cubic meters of water (Water Footprint, WF) and the percentage of food lost (Food Loss and Waste, FLW) arisen in the whole food chain.

Table 5.2: Environmental impact of one kilogram of each food category.

Food category	CF	WF	FLW
Cereals	0.64	1.81	0.38
Dairy & eggs	3.02	3.41	0.12
Fish & seafood	2.83	0.04	0.31
Fruit & vegetables	0.25	0.66	0.46
Meat	10.49	6.87	0.22
Oilseeds & pulses	1.26	2.11	0.20
Roots & tubers	0.14	0.20	0.52
Remaining food	0.11	5.39	0.32

Own elaboration from previous literature

Aiming to quantify the CF, WF and FLW associated to one kilogram of each food category from the food basket (Section 5.2.1), the estimations from previous literature have been contrasted and adapted following the classification in Table 5.A.1. In the case of the CF, information has been extracted from two sources. On the one hand, Aguilera et al. (2021) has been considered for *cereals*, *fruit & vegetables*, *meat*, *oilseeds & pulses*, *remaining food* and *roots & tubers*. The CF of these categories is calculated by dividing the added annual emissions of their subcategories by their production in year 2018 (FAO, 2021a). On the other hand, Forero-Cantor et al. (2020) have been used for the *fish & seafood* and *dairy & eggs* figures. Similarly, information on the WF has been taken from two sources: Yu et al. (2010) for *fish & seafood* and Blas et al. (2019) for the remaining categories. The former source provides an estimation of the WF generated by the UK fishery sector between 1999 to 2001. Therefore, it has been divided by the average production of this sector during those years (FAO, 2021b). The latter source allows to compute the WF of the remaining categories by calculating the total WF associated to their subcategories through the combination of their national and international WF and adding them. In the case of the FLW, the footprint for every food category is calculated using the percentages of FLW produced in each stage of the production process provided by Gustavsson et al. (2011) for the case of Europe incl. Russia. Table 5.2 presents the resultant environmental impacts associated to the consumption of one kilogram of each food category. As claimed in Funke et al. (2022), animal-sourced products, especially *meat*, are the largest producers of CO_2 equivalent emissions. With respect to WF, *meat* and *remaining food* require the largest amount of water cubic meters. However, the quantity of water associated to one kilogram of *dairy & eggs*, *oilseeds & pulses* and *cereals* is also noticeable. In the case of FLW, the largest loss and waste of food is generated with the *roots & tubers* and *fruit & vegetables* although the loss and waste generated in the remaining categories seems to be high as well.

Table 5.3: Environmental impact of the average monthly food basket in Spain from 2005 to 2021.

Food category	CF	WF	FLW
Cereals	2.63	7.45	1.57
Dairy & eggs	30.64	34.60	1.24
Fish & seafood	6.42	0.08	0.71
Fruit & vegetables	3.54	9.58	6.59
Meat	45.20	29.61	0.94
Oilseeds & pulses	0.95	1.58	0.15
Roots & tubers	0.36	0.50	1.32
Remaining food	1.81	86.76	5.08
Total	91.54	170.17	17.60

Table 5.3 shows the environmental impact associated to the *benchmark basket* based on the environmental impact of a kilogram from each food category. This Table shows that the average food consumption of an Spanish person between 2005 and 2021 produces 92 equivalent CO_2 kilograms, requires 170 cubic meters of water and wastes more than 17 kilograms of food each month. In particular, almost 90% of the total CO_2 equivalent emissions derive from animal-sourced products, over half of the water is used for products in the *remaining food* category and more than one third of food waste is produced in *fruit & vegetables*.

Table 5.4 details the changes in the three environmental dimensions studied derived from the variation observed in the average food basket between 2005 and 2021 (columns 2 and 4 in Table 5.1). Variations in the overall food basket have contributed to decrease CF in more than 12 kilograms and FLW in 200 grams. By contrast, WF has increased in 0.7 cubic meters. The decrease in the consumption of animal-sourced products, especially *meat*, has lead the CF reduction. This decrease in the consumption of animal-sourced products has pushed down the WF, but the increase in the consumption of products from the *remaining food* category has boosted the WF above it. The slight decrease in the FLW is mostly derived from the decrease in *cereals* consumption, being counter-balanced by the increase of *remaining food* consumption.

Table 5.4: Changes in the environmental impact from 2005 to 2021.

Food category	CF	WF	FLW
Cereals	-0.75	-2.12	-0.44
Dairy & eggs	-3.83	-4.32	-0.15
Fish & seafood	-1.47	-0.02	-0.16
Fruit & vegetables	0.05	0.15	0.10
Meat	-6.63	-4.35	-0.14
Oilseeds & pulses	0.12	0.20	0.02
Roots & tubers	-0.02	-0.03	-0.08
Remaining food	0.23	11.19	0.66
Total	-12.29	0.70	-0.21

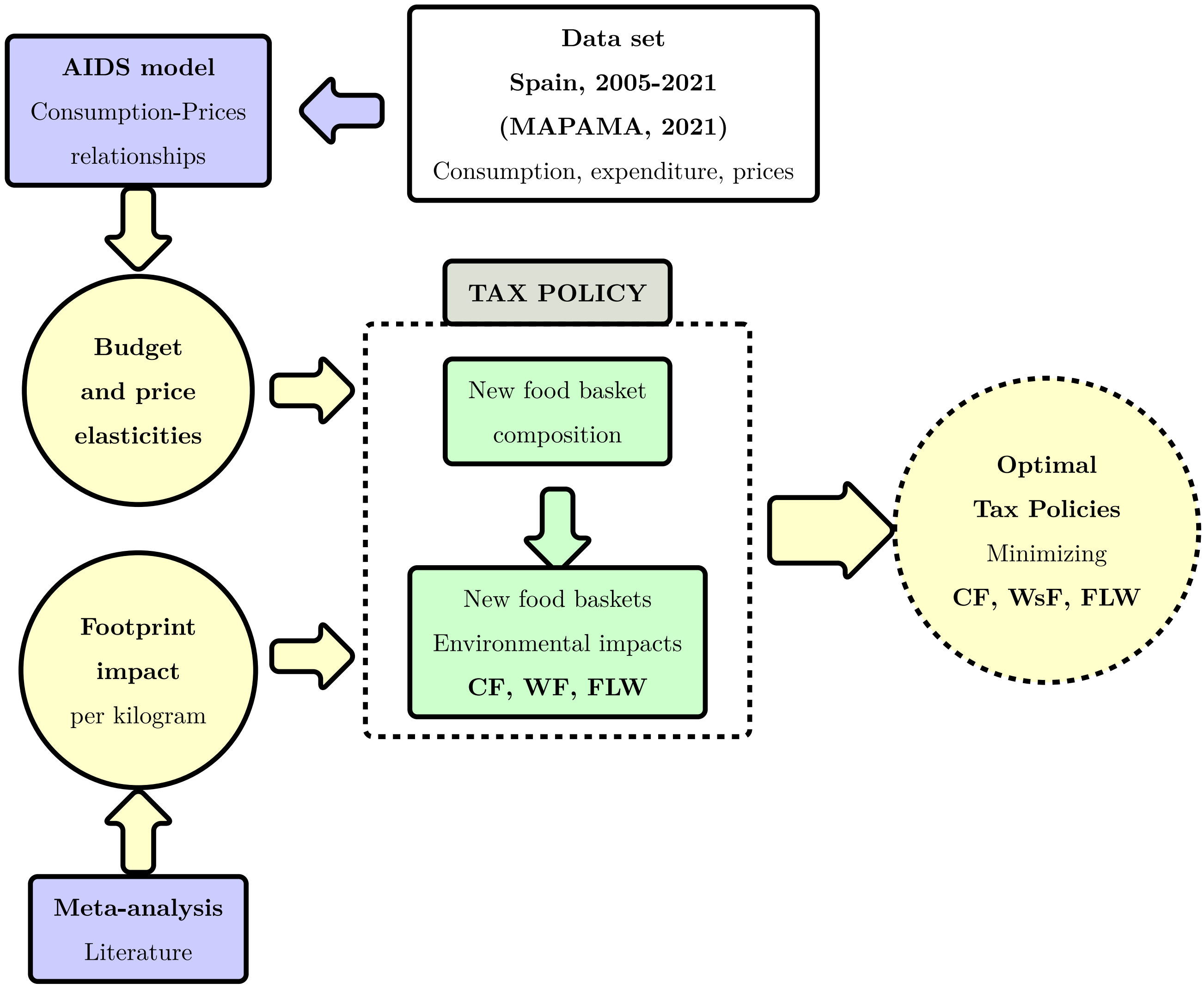

Figure 5.1: Representation of the methodological strategy. White, blue, green and yellow objects represent data source, tools, policies and outputs (intermediate and final), respectively.

5.3 Methodology

Aiming to contribute to the design of fiscal policies encouraging eco-friendlier consumption, the present analysis adopts a methodological strategy consisting on two steps. Firstly, as food taxes (subsidies) may be represented by positive (negative) variations in their prices, the relationship between food consumption and prices variations estimated through Almost Ideal Demand System (AIDS) (Deaton and Muellbauer, 1980a) is used as a representation of the variation in consumption derived from the implementation of unitary taxes (subsidies). Secondly, using the environmental impact associated to the consumption of each food category (Table 5.2) and the relationship between taxes (subsidies) and their consumption, the optimal fiscal policies to minimize the environmental impact of food consumption are searched using optimization problems with the COBYLA algorithm (Ypma et al., 2020; Powell, 1994). This methodological strategy is illustrated in Figure 5.1 and described in detail in the following subsections.

5.3.1 AIDS estimation

The AIDS (Deaton and Muellbauer, 1980a) defines the food demand of households in terms of food prices and total expenditure by using a functional form for the expenditure and utility of the consumers consistent with empirical household data. Among its advantages, AIDS is simple to estimate and the conditions imposed for homogeneity and symmetry can be tested through linear restrictions on fixed parameters. In addition, the utility function used may be aggregated over consumers to represent market demand.

In particular, the utility function specified in the AIDS (5.1) represents the minimum expenditure with which a consumer could reach a utility level (u) given the food prices (p).

$$\log(c(u,p)) = \alpha_0 + \sum_i \alpha_i \log(p_i) + \frac{1}{2}\sum_i \sum_j \gamma^*_{ij} \log(p_i)\log(p_j) + u\beta_0 \prod_i p_i^{\beta_i}, \tag{5.1}$$

where p_i corresponds to the price of food category $i = 1, \ldots, n$ and α_i, γ^*_{ij} and β_i are parameters. Given that the budget share allocated to a food category i (w_i) may be represented as the price derivative of the expenditure function $\left(w_i = \frac{p_i\, q_i}{c(u,p)} = \frac{\partial \log(c(u,p))}{\partial \log(p_i)}\right)$, Equation 5.1 allows to define a set of equations representing the budget share allocated by households to each food category in terms of their prices and the total budget (x) as follows:

$$w_i = \alpha_i + \sum_j^n \gamma_{ij} \log(p_j) + \beta_i \log\left(\frac{x}{P}\right), \tag{5.2}$$

where

$$\gamma_{ij} = \frac{1}{2}\left(\gamma^*_{ij} + \gamma^*_{ji}\right)$$

and P is the general price index defined as:

$$\log(P) = \alpha_0 + \sum_i^n \alpha_i \log(p_i) + \frac{1}{2}\sum_j^n \sum_i^n \gamma_{ij} \log(p_i)\log(p_j).$$

Adding-up $\left(\sum_{i=1}^n \alpha_i = 1, \sum_{i=1}^n \gamma_{ij} = 0, \sum_{i=1}^n \beta_i = 0 \;\forall j\right)$, homogeneity $\left(\sum_j \gamma_{ij} = 0 \;\forall i\right)$ and symmetry $(\gamma_{ij} = \gamma_{ji} \;\forall i, j)$ restrictions are derived from the previous specification of the AIDS. In particular, the adding-up restrictions allows to ensure that the sum of the budget shares allocated to each food category, as expected by their definition, is one. The homogeneity restrictions imply that consumers do not experience "money illusion". Therefore, the budget shares allocated to each food category remain constant unless the relative prices of the food categories or the total expenditure vary. By imposing symmetry, the Slutsky symmetry is satisfied.

The relationships of the consumption of each food category with the total budget (*budget elasticity*) and the price of food categories (*price elasticity*) cannot be directly obtained from the set of Equations 5.2. Instead, these elasticities are calculated using the estimations from the set of Equations 5.2 as follows (Buse, 1994; Green and Alston, 1990):

Budget elasticity:

$$\eta_i = 1 + \frac{\beta_i}{w_i}. \tag{5.3}$$

Price elasticity:

$$e_{ij} = \frac{1}{w_i}\left(\gamma_{ij} - \beta_i\left(\alpha_j + \sum_k \gamma_{kj}\ln(p_k)\right)\right) - \delta_{ij}, \tag{5.4}$$

where δ_{ij} takes value one when $i = j$ and zero otherwise.

The budget elasticity (η_i) measures the percentage variation in the consumption of food category i when the total budget experiences a variation of 1%, maintaining the other factors constant (*ceteris paribus*). If a budget elasticity is negative, it reflects that the consumption of the food category varies inversely to the budget of the consumer (i.e. the food category is an inferior good). By contrast, positive values of a budget elasticity reflect that the food category varies in the same direction as the budget of the consumer (i.e. the food category is a necessity). In particular, when the budget elasticity is larger than 1, consumption of the food category reacts more than proportionally to budget variations (i.e. the food category is a luxury good) (Frank, 2008).

The price elasticity (e_{ij}) quantifies the percentage variation in the consumption of food category i when the price of food category j is modified in 1%, *ceteris paribus.* Since variations in prices alter the purchasing power of the consumer and, consequently, her utility, the compensated price elasticities (e^h_{ij}) are also estimated to show what would be the percentage change in consumption with a unitary percentage change in prices holding the initial level of utility (Deaton and Muellbauer, 1980b). Using the Slutsky equation, the compensated elasticities can be calculated as follows (Green and Alston, 1990):

$$e^h_{ij} = \frac{1}{w_i}\left(-w_i\delta_{ij} + w_i w_j + \gamma_{ij} + \beta_i\beta_j \log\left(\frac{x}{P}\right)\right). \tag{5.5}$$

In the present analysis, variations in the consumption of a category with respect to its price (own-price elasticities) are estimated through uncompensated elasticities (Equation 5.4). If an own-price elasticity is lower than -1, the consumption of the food category is considered to be affected by its price (i.e. the demand of this food category is elastic). By contrast, if the own-price elasticity is larger than -1, the consumption is considered to be unaffected by the price (i.e. the demand of this food category is inelastic)(Frank, 2008). Variations in the consumption of a category with respect to other prices (cross-price elasticities) are estimated through compensated elasticities to consider the possibility that the consumer modifies her consumption looking for a decision that provides similar utility to the initial one. The sign of the cross-price elasticity indicates the relationship between the corresponding pair of food categories (Frank, 2008). In particular, negative values show that the consumption of the food category varies inversely to the price of the other category (i.e. the former food category is complement to the latter food category). Thus, increases (decreases) in the price of the latter category are associated to decreases (increases) in the consumption of the former food category. By contrast, positive values reflect that the consumption of the food category is directly affected by the changes in the price of the other category (i.e. the former food category is substitute of the latter food category). Consequently increases (decreases) in the price of the latter category are associated to increases (decreases) in the consumption of the former food category.

The results of this section have been estimated using the *micEconAids* package in R. In particular, AIDS have been estimated through Iterative Linear Least Squares (ILLS), which aims to reproduce a nonlinear estimation of the AIDS iterating linear estimations of it (Henningsen, 2017).

In practice, the serial correlation in prices and consumption series as well as their reduced size may difficult the estimation and inference of the AIDS model (Greene, 2012; Baltagi, 2008). Following Forero-Cantor et al. (2020), Moving Block Bootstrap (MBB) (Canty and Ripley, 2021; Davison and Hinkley, 1997) technique has been applied along with AIDS. MBB generates a given number of subsamples by concatenating equal-length pieces from the original time series in order to avoid serially correlated errors (Mizobuchi and Tanizaki, 2014). In this analysis 1000 subsamples are generated using pieces of 4 consecutive months from the original sample and then the AIDS model is estimated for each subsample.

Since the random generation of series in MBB differs depending on the initial point, MBB process has been repeated with 100 initial different points to achieve more robust results. Consequently, each elasticity is estimated and its statistical significance is tested using 100,000 estimations.

5.3.2 Optimization problem

In the present analysis, the optimal fiscal policy is considered to be the set of taxes and/or subsidies implemented on the different food categories that incentives the consumption pattern with the lowest environmental impact (i.e. CF, WF and FLW). In order to find this optimal fiscal policy, the following optimization problem is defined:

$$\min_{t_1,\, t_2,\, \ldots,\, t_n} \quad a \cdot \frac{CF(t_1,\, t_2,\, \ldots,\, t_n)}{CF(t_0)} + b \cdot \frac{WF(t_1,\, t_2,\, \ldots,\, t_n)}{WF(t_0)} + c \cdot \frac{FLW(t_1,\, t_2,\, \ldots,\, t_n)}{FLW(t_0)} \tag{5.6}$$

$$s.t. \; -20 \leq t_i \leq 20 \; for\, i = 1, \ldots n,$$

where i denotes each of the food categories with the exception of *remaining food* (which given the mix of products involved is not considered a taxable category) and t represents the tax rate applied to their prices.

The objective function of this problem has two components. The first component evaluates the performance of the taxes in reducing the environmental impact by comparing the environmental impact derived from the consumption with and without the taxes. $FP(t_1,\, t_2,\, \ldots,\, t_n)$, $FP = CF,\, WF,\, FLW$, denotes the environmental impact resulting from the application of the taxes whereas $FP(t_0)$ refers to the environmental impact before their implementation. If a set of taxes is successful in changing the consumption pattern towards a more environmentally-friendly one, the ratio of the corresponding environmental impact will take values lower than 1 as $FP(t_1,\, t_2,\, \ldots,\, t_n) < FP(t_0)$. Larger distance between the value of the ratio and 1 reflect a larger efficacy of the taxes in reducing the corresponding environmental impact. The lowest value attainable would be 0, corresponding to a case in which either there is no food consumption or consumption has no environmental impact. If a set of taxes modifies the consumption pattern towards less environmentally-friendly alternatives, the ratio of the corresponding environmental impact takes values larger than 1 as $FP(t_1,\, t_2,\, \ldots,\, t_n) > FP(t_0)$. Having no upward limit, greater magnitude of the ratio reflects that taxes induce consumers to worse food consumption alternatives in terms of environmental impact. If the set of taxes results in a consumption pattern with an environmental impact similar to the reached without them, the ratio of the corresponding environmental impact is 1 as $FP(t_1, t_2, \ldots, t_n) = FP(t_0)$. Since these ratia measure the performance of the taxes in each environmental dimension without relying on particular units of measurement, the global environmental performance of the taxes may be obtained by adding their performance in each dimension (Fang et al., 2014; Page et al., 2012).

The second component of the objective function represents the social preferences to minimize each environmental aspect studied. By giving higher (lower) values to coefficients a, b, c a stronger (weaker) preference is shown to decrease CF, WF and FLW, respectively. In this regard, the present analysis studies four different scenarios of social environmental preferences. The first scenario consists on a multidimensional approach that considers the three environmental dimensions defined (CF, WF and FLW) weighted according to their average magnitude from 2005 to 2021 ($a = 0.33$, $b = 0.61$, $c = 0.06$). The second scenario considers that the only environmental dimension with social interest is the CF ($a = 1$, $b = 0$, $c = 0$). The third scenario represents the case in which social preferences focus only on the WF dimension ($a = 0$, $b = 1$, $c = 0$). The fourth scenario corresponds to the case in which only the FLW dimension is of social interest ($a = 0$, $b = 0$, $c = 1$).

The final value of the objective function must be interpreted considering both components, the ratia of the environmental impact and the social environmental preferences. When the objective function equals the sum of the coefficients defining the preferences (a, b, c), the implementation of taxes does not modify the environmental impact derived from the food consumption without them. By contrast, when the value of the objective function is lower (greater) than the sum of the weights, the implementation of taxes is associated to an improvement (deterioration) of the environmental impact derived from food consumption.

The set of restrictions imposed in this optimization problem constrains the range of the taxes between -20 and 20% to be socially acceptable. While positive taxes are expected to disincentive the consumption of the corresponding food category by increasing its price, negative taxes or subsidies are expected to incentive the consumption of the corresponding food category by decreasing its price.

The optimization problem (5.6) is solved using *nloptr* package from R (Ypma et al., 2020) with the COBYLA algorithm (Powell, 1994). This algorithm explores the vertices of the region resulting from the linear approximation of the objective function and the constraints in order to find the values that minimize it.

5.4 Results

The willingness of consumers to modify their food consumption pattern when prices vary determines the efficacy of fiscal policies in changing their behavior and decreasing the environmental impact. In this respect, the elasticity of food consumption with respect to food prices plays an important role in the design and success of food taxes. Therefore, this Section summarizes how food consumption is affected by food prices before presenting the taxes that minimize the environmental impact for each scenario of social preferences defined previously.

5.4.1 Price elasticities of food consumption

Table 5.5 presents the budget and price elasticities of food consumption estimated through the AIDS. Values in the diagonal of the price elasticities represent own-price elasticities, which are calculated as uncompensated elasticities. Off-diagonal values represent cross-price elasticities, which are calculated as compensated elasticities. In order to interpret the change in consumption derived from the variation in the budget or a particular

Table 5.5: Elasticities of food consumption*.

		Price							
Consumption	Budget	Cereals	Dairy & eggs	Fish & seafood	Fruit & vegetables	Meat	Oilseeds & pulses	Roots & tubers	Remaining food
Cereals	0.75	-0.53		-0.53	0.60	0.60	-0.35		
Dairy & eggs	0.84		-0.37		0.46				
Fish & seafood	1.49	-0.32		-0.71	-0.26	0.80			0.44
Fruit & vegetables	0.77	0.26	0.35	-0.20	-0.41		0.21	0.17	-0.42
Meat	1.26	0.20		0.46		-0.65	-0.14	-0.21	
Oilseeds & pulses									
Roots & tubers									
Remaining food	0.61			0.25	-0.36		0.18		

* Blank spaces indicate that the elasticity is not significantly different from 0 at 5% significance level. Estimated elasticities are consistent with adding-up, homogeneity and symmetry conditions.

price, the rest of the variables are assumed to be constant. Consumption of each food category is differently affected by budget as well as by own- and cross- price variations. *Oilseeds & pulses* and *roots & tubers* are the two exempted categories which consumption seems neither determined by the budget nor the prices. The budget elasticities reflect that most of the remaining food categories could be considered necessities as their budget elasticity ranges between 0 and 1. However, *fish & seafood* and *meat* are luxury goods as their budget elasticity is above 1. Regarding own-price elasticities, the consumption of *cereals*, *fish & seafood*, *fruit & vegetables*, *meat* and *dairy & eggs* seems affected by its price. Being negative and larger than -1, the own-price elasticities of these categories reflect that their demand is inelastic. The inelastic demand of these categories coincides with the figures indicated in Forero-Cantor et al. (2020) and Cattaneo et al. (2021). With respect to cross-price elasticities, a variety of relationships arises. Complementarity is found between multiple categories, one of the most relevant for the optimal fiscal policies is the one of *fruit & vegetables* and *fish & seafood*. The relationships of substitution seem more present in the optimal fiscal policies. In particular, the most outstanding relationships are those of *meat* and *cereals*, *meat* and *fish & seafood*, *dairy & eggs* and *fruit & vegetables*, *remaining food* and *oilseeds & pulses*. Among these relationships, the complement of *fish & seafood* and *fruit & vegetables* as well as the substitution between *meat* and *fish & seafood* are documented in Forero-Cantor et al. (2020) and Gustavsen and Rickertsen (2013).

5.4.2 Optimal food taxes

Table 5.6 presents the optimal taxes minimizing the environmental impact from 2005 to 2021 associated to the food consumption of an Spanish person given the different scenarios of social preferences defined in Section 5.3.2 as well as their average accomplishment.

Several findings may be highlighted in Table 5.6. Firstly, optimal taxes are found in the extremes of the range established, -20 and 20%. Even though taxes may be considered the most straightforward tool to minimize the environmental impact by penalizing the consumption of most harmful food categories (Forero-Cantor et al., 2020), a combination of taxes and subsidies on the different food categories seems optimal for this target

Table 5.6: Optimal fiscal policies for different environmental preferences*.

		Unidimensional		
Food category	Multidimensional	Only CF	Only WF	Only FLW
Cereals	-20	-20	-20	-20
Dairy & eggs	20	20	20	-20
Fish & seafood	-20	-20	-20	20
Fruit & vegetables	20	-20	20	20
Meat	20	20	20	-20
Oilseeds & pulses	-20	20	-20	-20
Roots & tubers	20	20	20	-20
Objective function	0.9047	0.8239	0.8968	0.8845

* Solve Equation 5.6 for the different preference scenarios when taxes are between -20 and 20%. Multidimensional: $a = 0.33$, $b = 0.61$, $c = 0.06$, CF: $a = 1$, $b = 0$, $c = 0$ WF: $a = 0$, $b = 1$, $c = 0$, FLW: $a = 0$, $b = 0$, $c = 1$.

independently on the social environmental preferences considered.

Secondly, the optimal taxes to be implemented differ depending on the environmental target defined by the social preferences. Nevertheless, optimal fiscal policies may be explained by the relationships between the consumption and prices of the food categories (Table 5.5) and their environmental impact (Table 5.3). Thus, in order to minimize CF, subsidies should be implemented for *cereals*, *fish & seafood* and *fruit & vegetables* and taxes for the remaining categories. These subsidies help to decrease the consumption of *meat* and *dairy & eggs*, which are the largest producers of CF in the *benchmark basket.* Simultaneously, the consumption of these two categories is reduced by the tax in their consumption. To minimize WF, subsidies should be implemented for *cereals*, *fish & seafood* and *oilseeds & pulses* whereas the consumption of the remaining categories should be taxed. Subsiding the consumption of *cereals*, *fish & seafood* and *oilseeds & pulses* allows to decrease the consumption of *meat* and *remaining food*, which are some of the largest producers of WF in the *benchmark basket.* Additionally, the taxes on *meat* and *dairy & eggs* further decrease the consumption of these categories. A different pattern is observed for the minimization of the FLW as only *fish & seafood* and *fruit & vegetables* are taxed, being the rest of the categories subsidized. The taxes on *fish & seafood* and *fruit & vegetables* consumption decrease the consumption of the latter and *remaining food*, which produce the largest FLW in the *benchmark basket.* In addition, the subsidy implemented to the *oilseeds & pulses* reinforces the reduction of the *remaining food* consumption. Since the multidimensional preferences consist on a combination of the unidimensional ones, the optimal fiscal policy of the former seems a mix of the optimal fiscal policies of the latter.

Thirdly, the efficacy of the fiscal policies in reducing the monthly environmental impact generated by food consumption of an Spanish person depends on the target established. In particular, the largest accomplishment is observed when minimizing the CF and FLW, cases in which the environmental impact is decreased around 18 and 12%, respectively. Targeting the minimization of WF alone allows to decrease the environmental impact in around 11% whereas combining the social preferences as in the multidimensional scenario leads to the lowest reduction of the total environmental impact, slightly lower than 10%.

Robustness of the empirical results has been tested regarding the limits imposed to tax rates by setting them in -10 and 10% (see Appendix 5.B for detailed results). When taxes are constrained between -10 and 10%, the sign of the optimal taxes does not vary with respect to the observed in Table 5.6 for any of the preference scenarios analyzed. Furthermore, optimal solutions are also achieved at extreme values. As expected, the decrease in environmental impact achieved is smaller in every preference scenario analyzed. Indeed, the reduction in the environmental impact achieved ranges between 4 and 9% depending on the preferences.

From the point of view of economic policy, it is also relevant to quantify the benefits of designing a fiscal policy that considers all the food categories rather than just focusing on one of them. Technically, this consists on comparing the solution obtained of the optimization problem for the different environmental preferences with the resultant when the tax or subsidy is allowed for only one category. In the latter case, the set of control variables of the optimization problem is represented by $\{0, \ldots, t_i, \ldots, 0\}$, being $i \in (1, 2, \ldots, n)$ the food category selected for the policy design. Fiscal policies based on individual food categories are less effective in every of the environmental scenarios analyzed (see Appendix 5.C for detailed results). The efficacy of these optimal policies varies depending on the environmental scenario and the food category considered. However, the largest environmental impact reduction achieved in each scenario does not exceed the 5%. As expected by the price elasticities of the different food categories, the optimal fiscal policies consist not only on taxing the most harmful category of the *benchmark basket* regarding the environmental dimension analyzed. Instead, subsidies to substitute categories are advisable in some cases. This is the case of the 20% subsidy to *fish & seafood* for the multidimensional and WF scenarios.

5.5 Policy implications

Previous literature has already estimated how taxes on different food affect the production of greenhouse gas emissions and other pollutants based on a similar procedure to the one of this analysis. In particular, Säll and Gren (2015) estimate the decrease in greenhouse gas emissions, nitrogen, ammonia and phosphorus derived from the imposition of taxes on meat and dairy products for the case of Sweden. The authors set the value of the tax to compensate the emissions generated with each kilogram of food and estimate their effect on emissions using the variation in demand obtained from the price-elasticities of consumption estimated with AIDS. The estimation of the AIDS is performed in two steps to distinguish between aggregated groups of food (step 1) and the subcategories within each group (step 2). Their estimations show that reducing consumption of beef and pork contributes the most to reducing greenhouse gas emissions and pollutants. In Forero-Cantor et al. (2020), authors evaluate the change in greenhouse gas emissions when taxes on animal-sourced food are implemented in Spain. Based on the price-elasticities of food consumption estimated with AIDS, the authors estimate the change in food demand and the consequent variations in the carbon footprint occurred when different taxes are implemented. Their estimations conclude that taxes on fish are the most effective in reducing carbon footprint.

The present analysis broadens the perspective of previous literature by incorporating additional environmental dimensions affected by the food consumption as the use of water and the loss and waste of edible resources. Besides evaluating the environmental impact of certain food taxes, the present analysis seeks the combination of taxes that minimize the environmental impact allowing the simultaneous implementation of different subsidies and taxes on the major food categories composing the Spanish food basket. In this respect, the present analysis represents more accurately the food basket by considering additional food categories within it.

The findings from this analysis support that taxing *meat* is associated with decreases in the greenhouse gas emissions as in Säll and Gren (2015). Nevertheless, the optimal solution to minimize the CF consists on taxing *meat*, *dairy & eggs*, *oilseeds & pulses* and *roots & tubers* while subsiding *cereals*, *fish & seafood* and *fruit & vegetables*. In contrast to Forero-Cantor et al. (2020), this analysis finds that taxing *fish & seafood* is only optimal when minimizing the FLW. Indeed, to achieve a larger efficacy in minimizing FLW, the tax to *fish & seafood* must be accompanied by a tax to *fruit & vegetables* and subsidies to the remaining taxable categories (*cereals*, *meat*, *dairy & eggs*, *oilseeds & pulses* and *roots & tubers*). Given the relationships observed between the consumption and prices of the different food categories, setting a combination of taxes and subsidies that incorporates these relationships seems to be the optimal solution to minimize the environmental impact of the most harmful categories. Therefore, the first policy recommendation derived from this analysis consists on considering all the food categories in the design of fiscal policies minimizing the environmental impact rather than constraining to the most harmful category.

The dissimilar combinations of optimal taxes found for each scenario of preferences reflect that each environmental dimension requires from particular measures. In particular, the optimal fiscal policy to minimize CF consists on the subsidization of *cereals*, *fish & seafood* and *fruit & vegetables* and the taxation of the remaining categories. The fiscal policy to minimize WF is similar to this one. However, for this target, *fruit & vegetables* are taxed while *oilseeds & pulses* are subsidized. In the case of FLW, the optimal policy seems to deviate from the previous ones, taxing only *fish & seafood* and *fruit & vegetables* and subsidizing the remaining categories. When targeting simultaneously these three environmental dimensions, changes are not only observed regarding the design of the optimal fiscal policy, but a reduction of its efficacy with respect to the cases in which CF, WF and FLW are targeted separately is notorious. The heterogeneous contribution of food categories to each environmental dimension and the relationships observed across them harden the specification of taxes that minimize all the environmental consequences simultaneously. Therefore, the second policy recommendation derived from this analysis is the need for policymakers to clarify the environmental goals in the design of fiscal food policies.

Even though the increase in the value of taxes seems to increase proportionally the efficacy in minimizing the environmental impact, targeting the minimization of the environmental impact through an excessive increase in the food taxes could have serious social consequences as environmental fiscal policies affect the nutrition (Funke et al., 2022; Vázquez-Rowe et al., 2020) and income (Säll, 2018) of people. Therefore, the third policy recommendation derived from the present analysis consists on considering social aspects as the health and wealth of people in the design of environmental fiscal policies. In this sense, external shocks altering food prices must be also considered as they may affect both, the social conditions and the efficacy of the fiscal policies designed. For instance, in June 2022, the FAO Meat Price Index was 12.7% higher than the year before, mainly because of the war in Ukraine (FAO, 2022). If this price increase were transferred to the Spanish market and this were the only change in prices, the estimates of the elasticities (Table 5.5) allow us to roughly predict that the basket of the representative household would change between 2021 and 2022, reducing the consumption of *meat* by 8.3% ($\approx 12.7 \times (-0.65)$) and increasing that of *fish & seafood* and *cereals* by 11% ($\approx 12.7 \times 0.86$) and 7.6% ($\approx 12.7 \times 0.6$), respectively.

Consumer preferences constitute another social aspect to be considered in the design of fiscal policies as they can be of great help to policymakers. The estimated budget elasticities reveal that variations in the budget of the consumers entangle changes more than proportional in the consumption of *fish & seafood*, which represents a key category for the minimization of the three environmental dimensions analyzed, and *meat*, which is one of the current largest producers of CF and WF, *ceteris paribus*. Furthermore, the literature is extensive in related aspects such as consumers willingness to pay for sustainable practices in natural resources (Onofri et al., 2018), water resources (Halkos and Matsiori, 2014) or biodiversity conservation (Bhandari and Heshmati, 2010). And it is also true when it comes to analyzing explicitly consumer preferences for food products with low environmental footprints. In general terms, research has found that consumers show a positive attitude towards food products with reduced footprints but there is a large heterogeneity among consumers attitudes and the factors driven their decisions and the willingness to pay for food products with reduced CF (Rondoni and Grasso, 2021; Canavari and Coderoni, 2020; Liu et al., 2017; Koistinen et al., 2013), WF (Grebitus et al., 2015) and FLW (Obuobi et al., 2022; Gracia and Gómez, 2020).

5.6 Conclusions

Food consumption has a significant impact on the environment in terms of greenhouse gas emissions (Aguilera et al., 2021), water use (Blas et al., 2019; Mekonnen and Hoekstra, 2012, 2011) and food loss and waste (Vázquez-Rowe et al., 2020; Gustavsson et al., 2011). In particular, the average food consumption from 2005 to 2021 of a person in Spain is associated with the emission of 92 equivalent CO_2 kilograms, the use of more than 170 cubic meters of water and the waste of more than 17 kilograms of food each month. New alternatives to current food diets are required to prevent widening the human impact on the environment and slow down the advance of climate change (FAO et al., 2021).

Aiming to find more environmentally friendly consumption patterns, previous literature has compared the consequences of predominant food diets regarding multiple environmental aspects. In this respect, vegetarian diets have been found to be more responsible with the environment while providing nutrition of good quality (Esteve-Llorens et al., 2021; Blas et al., 2019; Castañé and Antón, 2017; Vanham et al., 2013). Fiscal policies have been proposed as a tool to incentive a shift in current consumption towards alternatives with lower environmental impact (Forero-Cantor et al., 2020; Säll and Gren, 2015). In order to accomplish this change, taxes are implemented to increase the price of environmentally damaging products whereas subsidies may be used to decrease the price of products more responsible with the environment. The present analysis aims to provide fiscal policy guidelines to achieve multiple environmental improvements in Spain complying with international commitments to mitigate climate change.

The success of fiscal policies on reducing environmental impact is mostly determined by the price elasticities of food consumption, which quantify the willingness of consumers to modify their consumption when prices vary. When consumption is poorly affected by prices (elasticity of consumption with respect to prices is low), fiscal policies need to implement larger taxes and subsidies to reach the desired variation in consumption. If consumption is not affected by prices (elasticity of consumption with respect to prices equals zero), fiscal policies will fail in achieving more environmentally friendly diets. Only when consumption is affected by prices (elasticity of consumption with respect to prices is high), fiscal policies will have the possibility to modify the behavior of consumers more easily (Frank, 2008). Following Forero-Cantor et al. (2020) and Säll and Gren (2015), the relationship between prices has been estimated through AIDS (Deaton and Muellbauer, 1980a). The estimations from this analysis reflect that the consumption of the food categories defined is inelastic with their own prices. Indeed, only the consumption of certain food categories (*cereals*, *fish & seafood*, *fruit & vegetables*, *meat* and *dairy & eggs*) is affected negatively by their own prices. By contrast, estimations show a larger elasticity of food consumption with respect to the prices of other food categories. Indeed, some of these elasticities are crucial in the design of the optimal fiscal policies. The substitution of *meat* with respect to *cereals* and *fish & seafood*, *dairy & eggs* with respect to *fruit & vegetables* and *remaining food* with *oilseeds & pulses* are the among the most noticeable in this aspect. Regarding complementary relationships, the one of *fruit & vegetables* with respect to *fish & seafood* may be highlighted.

Since the human impact on the environment is manifested in multiple ways (Fang et al., 2014), the environmental goal must be clearly identified in the design of fiscal policies. In this analysis, the environmental impact is measured in three dimensions: the greenhouse gas emissions, the water use and the food loss and waste. Thus, optimal fiscal policies are searched for the minimization of each dimension as well as for their combination. Solving the optimal problems for each environmental goal leads to four different combinations of optimal taxes and subsidies, each of them fitting best to the accomplishment of one environmental goal. The success in reducing human environmental impact also depends on the environmental aspects targeted. In general, implementing the optimal taxes within feasible limits, between -20 and 20%, allows to decrease the food consumption impact between 9 and 18% depending on the environmental goal considered. However, when taxes are constrained between -10 and 10%, the reduction of the environmental impact also halves, being between 4 and 9%.

Based on the main findings of this analysis, several policy implications can be derived. Firstly, fiscal policies that intend to address the environmental impact derived from food consumption should be designed taking into account all the food groups rather than focusing only on the most harmful group regarding the environmental dimension targeted. Secondly, the environmental goal must be clearly defined when designing policies for the mitigation of the climate change as each environmental aspect requires from different actions. Thirdly, food taxes may be implemented to enhance diets with a more responsible environmental use. Nevertheless, these changes entangle nutrition and wealth consequences that must be considered in the policy design.

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Appendix 5.A Classification of food subcategories into major food categories

Table 5.A.1: Classification of MAPAMA (2021) food categories.

Food category	Subcategories
Cereals	Bread, cereals, pasta, flour and semolina.
Dairy and eggs	Eggs (in kgs.), milk (liquid, reconstituted, condensed, powder and evaporated), milkshake, yogurt smoothie, butter, cheese, ice creams, cakes, cream, custard, flans, curds, chocolate cream, Catalan cream, cream desserts, other dairy products.
Fish and seafood	Fish (including canned and frozen).
Fruit and vegetables	Pickles, fruit and vegetables (including canned and frozen).
Meat	Meat (including canned and frozen).
Oilseeds and pulses	Legumes (including canned), olives and nuts.
Roots and tubers	Potatoes.
Remaining food	Honey, pastries, biscuits, chocolate, cocoa, sugar, sweeteners, oils, beverages, pizzas, tortillas, soups, spices, coffee, infusions, salt and other products.

Appendix 5.B Optimal fiscal policies with lower limits

Table 5.B.1: Optimal fiscal policies when taxes range from -10 to 10% *.

		Unidimensional		
Food category	Multidimensional	Only CF	Only WF	Only FLW
Cereals	-10	-10	-10	-10
Dairy & eggs	10	10	10	-10
Fish & seafood	-10	-10	-10	10
Fruit & vegetables	10	-10	10	10
Meat	10	10	10	-10
Oilseeds & pulses	-10	10	-10	-10
Roots & tubers	10	10	10	-10
Objective function	0.9524	0.9119	0.9484	0.9423

* Solve Equation 5.6 for the different preference scenarios when taxes are between -10 and 10%. Multidimensional: $a = 0.33$, $b = 0.61$, $c = 0.06$, CF: $a = 1$, $b = 0$, $c = 0$ WF: $a = 0$, $b = 1$, $c = 0$, FLW: $a = 0$, $b = 0$, $c = 1$.

Appendix 5.C Efficacy of fiscal policies focused on one food category

Table 5.C.1: Optimal fiscal policies when food categories are taxed individually*.

			Unidimensional					
	Multidimensional		Only CF		Only WF		Only FLW	
Food category	Tax	Objective function	Tax	Objective function	Tax	Objective function	Tax	Objective function
Cereals	-20	0.9918	-20	0.9862	-20	0.9949	-20	0.9903
Dairy & eggs	20	0.9871	20	0.9776	20	0.9887	-20	0.9791
Fish & seafood	-20	0.9692	-20	0.9686	-20	0.9652	20	0.9895
Fruit & vegetables	20	0.9962	-20	0.9736	20	0.9833	20	0.9643
Meat	20	0.9739	20	0.9505	20	0.9827	-20	0.9898
Oilseeds & pulses	-20	0.9955	20	0.9867	-20	0.9873	-20	0.9820
Roots & tubers	20	0.9911	20	0.9806	20	0.9946	-20	0.9895

* Solve Equation 5.6 for the different preference scenarios when taxes are between -20 and 20% and policies are only based in one food category. Multidimensional: $a = 0.33$, $b = 0.61$, $c = 0.06$, CF: $a = 1$, $b = 0$, $c = 0$, WF: $a = 0$, $b = 1$, $c = 0$, FLW: $a = 0$, $b = 0$, $c = 1$.

Chapter 6

Distributional impact of COVID-19: Regional inequalities in cases and deaths in Spain during the first wave[1]

Spain is being hit hard by the COVID-19 pandemic. During the first wave, from mid-March to early June 2020, the disease caused nearly 30,000 deaths in a population of 47 million. This article quantifies the unevenness in the distribution of epidemiological variables across the Spanish territory. The study is relevant because Spain is divided into regions that hold devolved authority for providing health care services to their citizens. Using inequality metrics, the study shows: i) By mid-April inequality in the epidemiological variables reached a stationary value that changed little with the incorporation of new cases and deaths. At the end of the outbreak, cumulative cases and deaths were fairly unevenly distributed across Spanish provinces; ii) Inequality shows a monotonic downward trend throughout the outbreak showing a decrease from the onset to the end ranging from 22% to 49% in cases and between 17% and 42% in deaths; iii) Over 90% of the inequality observed can be attributed to differences between regions, while less than 10% is due to the differences across provinces within regions. Awareness of the existence and nature of the inequality observed in the epidemiological variables is needed to develop successful policies to improve health services in Spain.

6.1 Introduction

The World Heath Organization declared COVID-19 disease as a pandemic on March 11, 2020 (WHO, 2020c) by which time there were 124,101 confirmed cases and 4,583 deaths, mostly in China. Since then, the virus spread very fast all around the world with more than 84 million confirmed cases and 1.8 million deaths as of January 7th, 2021 (WHO, 2020a).

Spain is one of the countries hardest hit by the pandemic with 1,893,502 confirmed cases and 50,442 deaths (data reported on January 7th, 2021, WHO (2020a)), in a population of about 47 million. It was one of the countries most affected by the first wave. At the beginning of July 2020, when the first wave was considered controlled, figures ranked Spain as fifth in the world in terms of deaths by population behind San Marino, Belgium, Andorra and the United Kingdom (data reported on January 7th, 2021, WHO (2020a)).

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However, the effects of the pandemic have been felt very unevenly across Spain. The first significant outbreaks classified as non imported cases appeared in early March in the capital, Madrid, which lies in the center of the country, but also to a lesser extent in the provinces of Araba/Álava and La Rioja, which lie close to each other in the north. From March 9 to 11, the regional governments of these regions imposed social isolation measures such as the suspension of school classes, the closure of universities, restrictions on visits to nursing homes and a ban on large-scale events.

By March 11 the cumulative figures for cases in Spain was over 4,400, more than 2,900 of them in Madrid. The rest were distributed very differently across the other 51 provinces. Only in three provinces (Barcelona, La Rioja and Araba/Álava) was the number of cases slightly above 200, and 22 provinces had no cases at all. However, it was not until March 14 that the Spanish government enacted the State of Alarm (*Estado de Alarma*) (BOE, 2020a). This legislation allowed the government to restrict the mobility of citizens and limit economic activity to essential sectors. In fact, during the first month of the State of Alarm, people remained in strict confinement at home, except for essential activities such as visiting doctor, basic shopping or work. The lockdown of the economy from March 29 to April 13, which restricted activities to essential sectors, had positive results in terms of flattening the epidemic curve (Saez et al., 2020; Tobías, 2020). The initial State of Alarm was extended subsequently several times in Parliament ending on June 21st, though some of the initial restrictions were lifted (BOE, 2020b,c,d,e,f,g). Recent studies estimates that these measures prevented up to 450,000 deaths in Europe (Flaxman et al., 2020). In the case of Spain, the Spanish National center for Epidemiology (SNCE, Instituto de Salud Carlos III (2020)) considered that the peak of the first outbreak was under control by the end of May and established a new surveillance and control strategy of the disease.

Not surprisingly, most current studies on Covid-19 focus on the trends in the epidemiological variables such as confirmed cases, hospital admissions and deaths. Modelling and analysing trends helps to assess where the pandemic is and to predict its future evolution (Ceylan, 2020; Liu et al., 2020). There are also studies that analyze the link between these variables and others of a socioeconomic nature, looking for significant links that may help to manage future pandemics. In this context, for the case of Spain there are analyses focusing on the links between the propagation of Covid-19 and variables such as the mobility of citizens (Aleta and Moreno, 2020; Mazzoli et al., 2020), local climate characteristics (Briz-Redón and Serrano-Aroca, 2020; Ma et al., 2020; Oto-Peralías, 2020; Paez et al., 2020), pollution concentration (Martorell-Marugán et al., 2021; Ogen, 2020) and the implications of enforced isolation on the evolution of the disease (Casares and Khan, 2020; Flaxman et al., 2020; Henríquez et al., 2020; Hyafil and Morina, 2020; Moosa, 2020; Siqueira et al., 2020; Zambrano-Monserrate et al., 2020).

Despite the interest shown in trends in epidemiological variables, research literature has so far paid little attention to distributional issues associated with the evolution of the epidemic variables. Such analysis are also relevant in understanding how local characteristics and measures taken may affect the evolution of the pandemic. The Spanish National Health System is based on a universal coverage, public funded, with a free of charge provision and some co-payment of pharmaceuticals related to age and income (Mentzakis et al., 2019). However, Spain is divided into 17 regions, which hold devolved authority for organizing and providing health care services to their citizens. All regional governments have been responsible for health care planning, organization and management since 2002, and are thus politically accountable to their constituents (Costa-Font and Moscone, 2009). In 2018, 92.6% of the public health expenditure were executed by the regions (Rodríguez Blas, 2020).

This status quo was modified with the initial State of Alarm, which included measures to centralize health services at national level (BOE, 2020a). The subsequent extensions of the State of Alarm shifted towards a co-governance system, with the central government setting the benchmark for actions and the regional governments organizing those actions within their regions. Ensuring coordination between the national and regional governments was one of the key points for the resilience of the Spanish health system during the early weeks of the pandemic (Legido-Quigley et al., 2020).

Distributional concerns can be quantified using inequality metrics. This is a standard methodology that has been applied to several topics, especially in the social sciences. A non-exhaustive list includes studies on the distribution of variables as diverse as income (Bui et al., 2017; Chongvilaivan and Kim, 2016; Ram, 2015), health resources (Saito et al., 2020; Morita et al., 2018; Alcalde-Unzu et al., 2009; De Maio, 2007), education facilities (Quadrado et al., 2001), sports results (Borooah and Mangan, 2012), demographic behavior (Bleha and Ďurček, 2019; Pagliacci, 2019), natural resources use (Cetrulo et al., 2020; Tang et al., 2020; Gutiérrez and Inguanzo, 2019; Duro et al., 2018) and pollutant emissions (Bolea et al., 2020; Pakrooh et al., 2020; Xia et al., 2019).

Our research uses this methodology to study how unevenly epidemiological variables of the first wave of Covid-19 were distributed across the provinces and regions of Spain. We focus on two aspects in particular. First, the distribution of cumulative cases and deaths across Spain's provinces from March to June 2020 is quantified using inequality indices. Second, the analysis seeks to learn whether the inequality observed in the distribution of cases and deaths is due to differences between the Spain's regions, which hold devolved authority to manage their health systems, or to differences within regions, reflecting idiosyncrasies of provinces, which may be affected by their population density, city sizes, airports, aging population, etc. This decomposition provides highly valuable information for policymakers. To address this second issue, we use the properties of the Theil index, which enables inequality to be decomposed into different levels (Shorrocks, 1984, 1980). The distributional analyses implemented in this study weight cases and deaths in provinces by their populations.

The rest of the paper is structured as follows. Section 2 describes the Spanish National Health System and the role of the central and regional governments. Sections 3 and 4 detail the data and the methodology used in our analysis. Section 5 starts by looking at the main initial messages that can be drawn from the data. In particular, it overviews the trends in confirmed cases and deaths, their relationship with population figures and how they are distributed across the different regions and provinces. After the context is analyzed, the trend in inequality indexes and the Theil decomposition are presented. Finally, Section 6 discusses the results and presents the conclusions.

6.2 The Spanish National Health System

The Spanish National Health System (SNHS) is a public funded health system based on universal coverage with free access to health care for almost all citizens and some co-payment of pharmaceuticals related to age and income. All residents in Spain have the right to full health coverage, regardless of their nationality and legal status. This right was only limited during 2012-2018 when the legislation in force linked the right to the legal and employment situation of the people, excluding, in practice, only undocumented immigrants from coverage (RDL 16/2012 (BOE, 2012) was repealed by RDL 7/2018 (BOE, 2018)) .

The SNHS is settled on the territorial organization of Spain established after the approval of the 1978 Constitution. Since then, Spain is divided administratively into 52 provinces grouped into 17 regions (called

Figure 6.1: Territorial organization of Spain: regions. Source: Reproduced from the Minister of Health, 2020.

"Autonomous Communities") and two autonomous cities located in the north of Africa. These regional and provincial divisions correspond to the NUTS 2 and NUTS 3 classifications respectively, as used by the European Union for statistical matters (Eurostat, EU, 2020b). Figure 6.1 represents this territorial division.

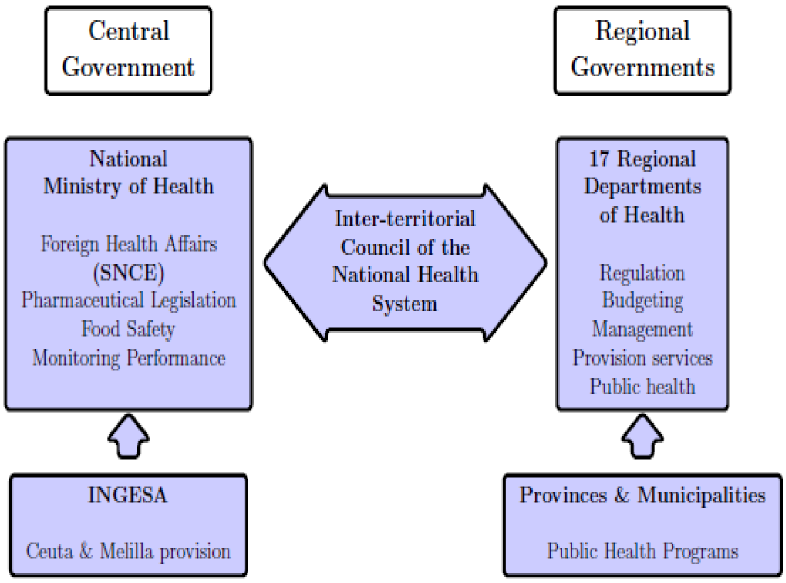


Figure 6.2: The decentralized approach of the Spanish National Health System.

From the organizational point of view, the SNHS is fully decentralized since 2002, with each of the 17 regional authorities being competent for the regulation, planning, budgeting, organization and management of health care within its jurisdiction, including the implementation of public health policies. In this decentralized framework, the national Ministry of Health acts as the guarantor of the equitable functioning of health services throughout the country. However its responsibilities are reduced to basic legislation and general coordinator in topics such as foreign health affairs (including those related to epidemiological control and fight against communicable diseases through the SNCE), pharmaceutical legislation, food safety and monitoring of health system performance. The national Ministry of Health is also responsible for the provision of services in the two autonomous cities located in the north of Africa, Ceuta and Melilla, representing 0.36% of the Spanish population. This provision is centrally managed through the Institute for Health Care Management (INGESA, in Spanish). Coordination between the national and regional public health administrations is carried out through the Inter-territorial Council of the national health system (CISNS) made up of 17 regional health ministers chaired by the national minister. Decisions of CISNS are expressed as recommendations that are adopted by consensus (see Bernal-Delgado et al. (2018) for a more in-depth description). Finally, provinces and municipalities do not play any relevant role in the decision-making of the SNHS. However, they do collaborate with regional health departments on public health programs. Figure 6.2 synthetics the SNHS organizational framework.

The initial State of Alarm caused by Covid-19 altered the regular operation of the SNHS. To cope with the outbreak and contain its spread, the management of public health policies was centralized within the Ministry of Health. This centralization allowed a more efficient purchase of necessary goods and services in the international markets and organization of the production of these goods at national level. This was specially relevant to increase work safety because shortages in personal protective equipment have been deemed one of the reasons for the number of medical staff infected in Spain during the first days of the outbreak (Henríquez et al., 2020). After the relaxation of the lockdown measures imposed by the State of Alarm, the co-governance system between the national and regional public health administrations re-emerged, with the national Ministry setting the basic strategies to fight the pandemic and the regional governments in charge of implement these strategies through the agreements reached within the CISNS.

Regarding financing, the SNHS is mainly funded by taxes. In general terms, the responsibility on tax collection is shared by the central and regional governments. The central government collects VAT, personal income tax and excise taxes, and afterwards, each regional government is assigned 50% of the personal income tax as well as 50% of the revenues generated within their territories by VAT and 58% of those yielded by selected excise taxes such as alcohol, tobacco and hydrocarbons. Tax rates for VAT and excise duties are common in all regions; however, the marginal rates of personal income tax can be revised by each region, within certain established limits. In addition, regional governments have full regulatory capacity on other taxes such as wealth, property, inheritance, gambling or car registration. Since Bernal-Delgado et al. (2018) point out, these resources grant to the regions a significant fiscal autonomy. This fiscal autonomy is even greater in the case of the Basque Country and Navarre, which collect all the taxes within their jurisdiction and then transfer a part to the central government in payment for the services provided in those regions.

In addition to taxes, the regions are also financed through the so-called Fund for Basic Public Services, whose objective is to guarantee a minimum level of basic public health services throughout the country (as well as educational and social services). Each region contributes 75% of its tax revenues to the fund, and then the central administration distributes it among all the regions according to a formula that considers the characteristics of the population, geographic extension, density and insularity. The system is complemented by other funds that aim to reduce the funding imbalance across the regions.

It is worth mentioning that none of the revenues indicated are specifically earmarked for health spending, but rather for financing all the services provided by the regions. In practice, this means that regional health systems can become very different from each other due to the internal allocation that each region makes of its funds among the various services it offers to its citizens. As an example of these differences Table 6.1 shows

Table 6.1: Regional public expenditure on health (2018).

Region*	GDP (in %)	Euros per capita
Andalucía (Andalusia)	6.3%	1,212
Aragón	5.7%	1,601
Asturias [Principado de ...]	7.4%	1,676
Canarias (Canary Island)	6.7%	1,399
Cantabria	6.5%	1,543
Castilla y León	6.6%	1,577
Castilla-La Mancha	7.1%	1,438
Cataluña (Catalonia)	4.7%	1,432
Comunitat Valenciana	6.3%	1,415
Extremadura	8.7%	1,626
Galicia	6.4%	1,491
Islas Baleares (Balearic Islands)	5.1%	1,407
La Rioja	5.4%	1,477
Madrid [Comunidad de ...]	3.6%	1,274
Murcia [Región de ...]	7.4%	1,567
Navarra (Navarre)	5.3%	1,651
País Vasco (Basque Country)	5.3%	1,753

Source: Rodríguez Blas (2020)

* In brackets the full name and in parentheses the English name of the region (where applicable)

regional public expenditure on health as percentage of regional GDP and in euros per capita.

6.3 Description of data

During the pandemic, by delegation of the national Ministry of Health, the SCNE has been in charge of monitoring the COVID epidemic (Instituto de Salud Carlos III, 2020). For the follow-up to be successful, a proper and rapid data collection is considered essential. However, during the first wave, the SNCE reported data on epidemiological variables at the regional level but not at the provincial level.

In all cases, the confirmed cases and deaths reported correspond to patients with positive PCR tests. The definition of positve PCR test varied throughout the first wave. Until May 10, all available laboratory techniques (PCR, ELISA serological test, rapid antibody test or antigen test) were considered. As of May 11, confirmed cases diagnosed by PCR or antigen technique are counted according to the strategy for early detection, surveillance and control of COVID-19 of the Ministry of Health, which was agreed by the technicians of all regions in the CISNS (Instituto de Salud Carlos III, 2020).

It is worth noting that many cases went undetected, especially at the onset of the outbreak, because either they were asymptomatic or the health system only tested more severely affected patients (Hyafil and Morina, 2020). Similarly, the data may had undercounted the true number of deaths due to the lack of PCR test to all deaths (Alamo et al., 2020). On the other hand, recent literature has cast some doubts on the epidemiologic pertinence of using PCR test if individuals are positive but not contagious (Jefferson et al., 2021).

The data on epidemiological variables at provincial level used in this research come from the EScovid19data public repository (EScovid19data, 2020) that was accessed on August 3, 2020. It runs under a Creative Commons Attribution 4.0 International license (CC BY 4.0). This repository provides the data reported by the SNCE for single-provincial regions. For the rest of the regions, the repository was updated daily by volunteers during the pandemic, who extracted and homogenized data mostly from the official regional health services. In general terms, each region or province had a sponsor who was responsible for obtaining data that could be downloaded automatically or, if that was not possible, for uploading it to a common spreadsheet. More details on how the repository works are available in its own web page (https://github.com/montera34/escovid19data). This repository is one of the open-data resources considered as pertinent for studing COVID-19 in Spain (Alamo et al., 2020) and has been used for academic works such as Martorell-Marugán et al. (2021), Briz-Redón and Serrano-Aroca (2020) and Paez et al. (2020). Table 6.A.1 in Appendix 6.A lists the Spain's regions with the corresponding provinces and their main source of information used by the repository.

With respect to population, 2019 data from the Spanish Statistical Institute (INE) is used.

6.4 Methodology

6.4.1 Measuring inequality

The simplest way to analyse the extent to which the distribution of a variable within individuals from a sample (provinces in our case) deviates from perfectly equal distribution is to draw a Lorenz curve (Lorenz, 1905). In our case, this curve relates the cumulative proportion of provinces weighted by population to the cumulative proportion of cases (or deaths), assuming that provinces are arranged in increasing order of cases (or deaths). A completely uniform distribution is shown by a diagonal line that represents the situation in which all provinces have the same number of cases or deaths given their population. The nearer the curve of the distribution is to this diagonal line, the more uniform the distribution is. One advantage of using a Lorenz curve to assess evenness is that it enables the distribution of a variable to be compared over time. When the Lorenz curves of two distributions are displayed in the same graph and do not cross, it can be stated unequivocally that the curve closer to the diagonal represents a more egalitarian situation than the other.

Apart from the graphic analysis provided for the Lorenz curve, the evenness of a distribution can be measured via inequality indexes. In general terms, an inequality measure is a function that ascribes a value to a specific distribution such that direct quantified comparisons can be made across different distributions. An inequality index is considered appropriate if it satisfies four basic properties: anonymity, population invariance, scale invariance and the Pigou-Dalton Transfer (Cowell, 2009). Among the inequality indices that hold these properties, for the purpose of this study, we use the Gini index (Gini, 1911) and the Theil index (Theil, 1967) which are some of the most widely used in social science.

The Gini index is inextricably linked to the Lorenz Curve because quantifies the degree of inequality of a distribution as the normalised area between the Lorenz curve of the distribution and the 45-degree line (line of perfect equality). Formally, the Gini index for the distribution of an epidemiological variable e among M provinces can be calculated as

$$G = \frac{1}{2\,e\,p^2} \sum_{i=1}^{M} \sum_{j=1}^{M} p_i\, p_j \; |e_i - e_j| \,, \tag{6.1}$$

where e_i and p_i represent the epidemiological variable (cases or deaths) and population of province i for $i = 1, 2..., M$, respectively. $p = \sum_{i=1}^{M} p_i$ is the overall population, and $e = \sum_{i=1}^{M} \frac{p_i}{p} e_i$ denotes the total of the epidemiological variable. This index ranges between 0 (maximun equality) and 1 (maximun inequality).

The main drawback of the Gini index is that it is neither easily decomposable nor additive. In addition, it does not respond in the same way to income transfers between people at opposite tails of the income distribution as it does to transfers in the middle of the distribution (Allison, 1978; Atkinson, 1970).

The Theil index belongs to the General Entropy family of indices, which are based on the notion of entropy in information theory (Theil, 1967). This family is expressed in terms of a parameter that expresses the sensitivity of the indicator to different parts of the distribution. The Theil index corresponds to the case in which this parameter takes a value of 1 meaning that all points in the distribution are treated equivalently. Formally, the Theil index is calculated as

$$T = \sum_{i=1}^{M} \frac{p_i}{p} \, \frac{e_i}{e} \, \ln\left(\frac{e_i}{e}\right). \tag{6.2}$$

The Theil index ranges between 0 (maximum equality) and $Ln(M)$, with M being the number of provinces in our study, (maximum inequality).

Notice also that the Theil index requires logarithms to be applied to the epidemiological variables. This is an important point for our analysis, since some provinces have zero cumulative cases and deaths in the early days of the outbreak. Following the advice of Bellù and Liberati (2006), we consider a value equal of 10^{-100} for these cases to solve this shortcoming.

Unlike the Gini index, the Theil index displays the property of additive decomposability, defined by Shorrocks (1984), which enables inequality to be decomposed by population sub-groups and expressed as a weighted sum of a within-group and a between-group component. This point is developed in the next subsection.

6.4.2 Regional decomposition of inequality

When a population can be partitioned into excluding subgroups, it is useful to decompose the dissimilarities observed by population sub-groups, expressed as a weighted sum of a within-group and a between-group component. The *within* component accounts for inequality inside each group and the *between* component accounts for inequality across groups. This is the case here, where data are available at provincial level and provinces can be classified by regions. Given that regions are in charge of the health services, we are interested in learning what part of the inequality observed is due to differences *within* and *between* regions.

The Theil index is one of the inequality measures that enables inequality to be decomposed additively between and within groups (Shorrocks, 1984, 1980). When applied to our study, the decomposition of the Theil index for the distribution of an epidemiological variable e among M provinces distributed among R regions, can be formally expressed as

$$\mathrm{T} = \mathrm{T}_{within} + \mathrm{T}_{between}, \tag{6.3}$$

being

$$\mathrm{T}_{within} = \sum_{r=1}^{R} \frac{\sum_{i=1}^{M_r} p_{i,r} \cdot e_{i,r}}{\sum_{i=1}^{M} p_i \cdot e_i} \cdot \mathrm{T}_r, \tag{6.4}$$

$$\mathrm{T}_{between} = \sum_{r=1}^{R} \frac{\sum_{i=1}^{M_r} p_{i,r} \cdot e_{i,r}}{\sum_{i=1}^{M} p_i \cdot e_i} \cdot \left[\ln \left(\frac{p}{p_r} \cdot \frac{\sum_{i=1}^{M_r} p_{i,r} \cdot e_{i,r}}{\sum_{i=1}^{M} p_i \cdot e_i} \right) \right], \tag{6.5}$$

where M_r is the number of provinces in region r, $p_{i,r}$ is the population of province i in region r and T_r is the value of the Theil index calculated with the population of region r alone.

The term $T_{between}$ reflects inequality due to differences observed between regions, while the term T_{within} represents inequality due to differences observed within the provinces of those regions. It is worth noting that the contribution of region r to total inequality, T, is given by $(\sum_{i=1}^{M_r} p_{i,r} \cdot e_{i,r} / \sum_{i=1}^{M} p_i \cdot e_i) T_r$. This term refers to the inequality *within* region r.

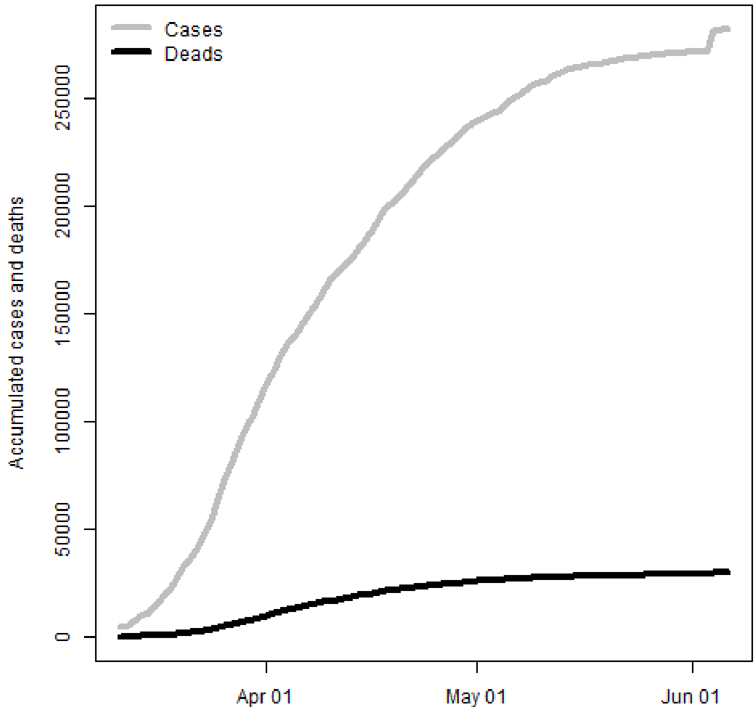
6.5 Results

6.5.1 Trend in cases and deaths

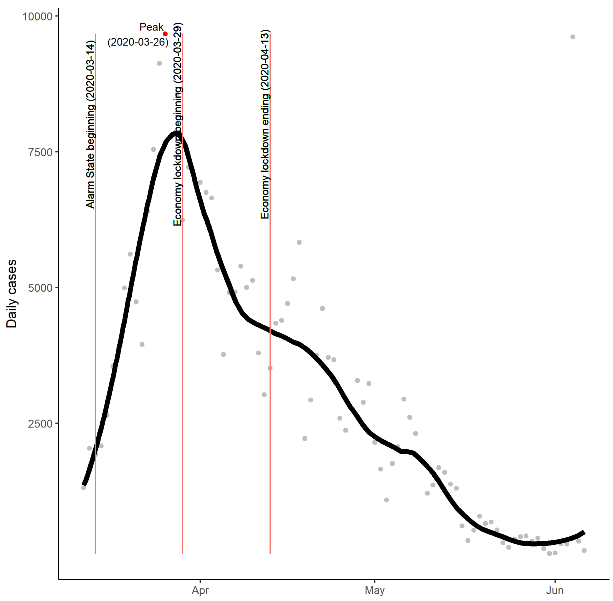
We focus our analysis on a period of interest determined by the curves of incidence and prevalence of the epidemic. In particular, the onset (ending) is set as the first (last) day on which the number of deaths was above (below) 30; that is from March 11 to June 6, 2020. Choosing this study period ensures that the data represent homogeneous information for the entire first COVID wave.

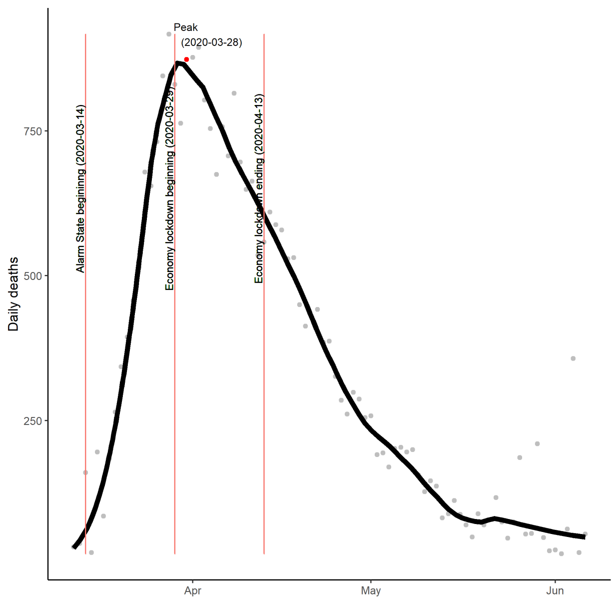
Time series for the epidemiological variables, cases and deaths are noisy, reflecting administrative lags in incorporating new information. In fact, weekend data show unreal reductions as data are reported at the start of the next week. Following Mazzoli et al. (2020) we eliminate this effect by smoothing the time series running average of three days assigning the value to the mid point.

Figure 6.3 shows the cumulative and numbers of cases and deaths per day for first Spanish wave. Cases per day are calculated as the interday variation of the cumulative data reported. At country level, daily cases and deaths peaked on March 26 and 28, 2020, respectively. These peaks are very close together taking into account that, based on China data, the WHO reported that the time between symptom onset and death ranged from about 2 to 8 weeks (WHO, 2020b). This confirms the idea that during the first part of the outbreak in Spain,



(a) Prevalence (Cumulative data)



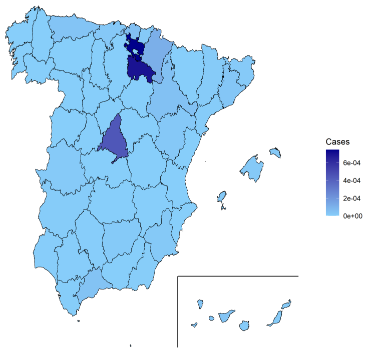


(b) Cases per day (points). Line: Estimated trend with local regression (loess) for $\alpha = 0.3$ (smoothness parameter) and $\lambda = 2$ (degree of the local polynomial).

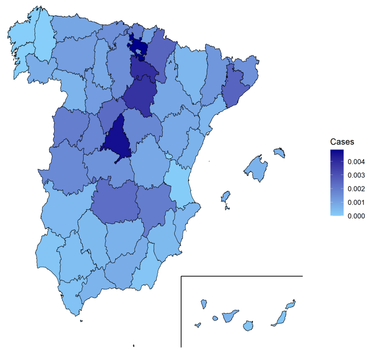
(c) Deaths per day (points). Line: Estimated trend with local regression (loess) for $\alpha = 0.3$ (smoothness parameter) and $\lambda = 2$ (degree of the local polynomial).

Figure 6.3: Epidemiological curves.

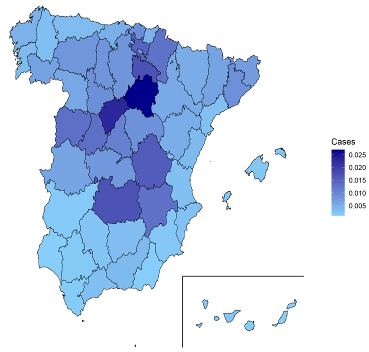
the fatality rate (deaths/cases) was significantly high, probably due to the pressure on health services (Verelst et al., 2020).



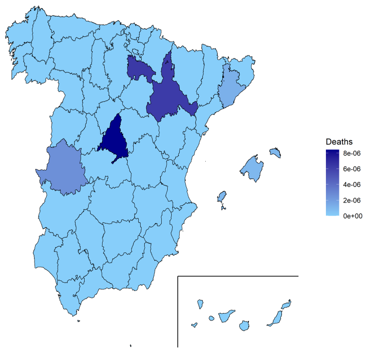
(a) Cumulative cases at the onset



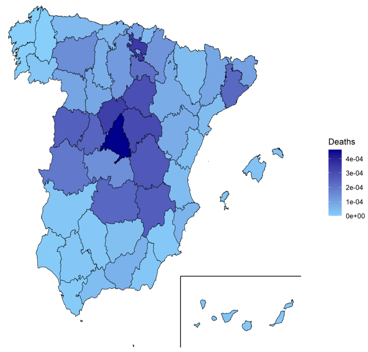
(b) Cumulative cases on the peak day



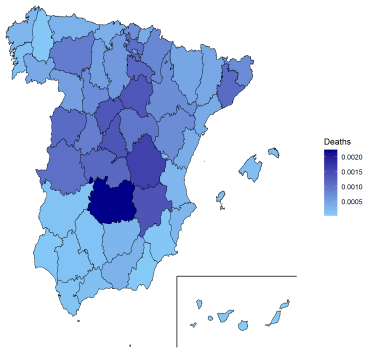
(c) Cumulative cases at the ending



(d) Cumulative deaths at the onset



(e) Cumulative deaths on the peak day



(f) Cumulative deaths at the ending

Figure 6.4: Distribution across provinces of per capita epidemiological variables on key dates.

At a first glance, the data shows epidemiological variables as heterogeneously distributed across Spain's provinces. Figure 6.4 shows the cumulative number of cases and deaths with respect to the population of each province on three key dates: at the onset of the disease, on the peak day and at the ending. Several facts deserve to be highlighted. First, except at the onset, a highly positive relationship is observed between the distribution of cases and the distribution of deaths across provinces. In fact, the Pearson correlation between cases and deaths across provinces is 0.89 for the peak day of cases, and 0.85 for the peak day of deaths. This means that the fatality rates between provinces are very similar over time. Second, there are differences in terms of rankings between provinces when the distributions are compared over the key dates. At the onset of the pandemic, the numbers of cases and deaths were low and concentrated in a few provinces reflecting local outbreaks in Madrid (in the center) and La Rioja (in the north). By the peak days the virus had already spread throughout the country, with Madrid and the surrounding provinces especially hard hit. By the ending the virus had spread more evenly across the provinces, but cases and deaths were still concentrated in the provinces in the center of the peninsula, while those on the coast were less affected. However, Madrid was no longer at the top of the ranking: it had been surpassed by Ciudad Real, one of its neighboring provinces belonging to Castilla La Mancha region, with a high percentage of elderly population. Third, the data trend also shows the role of Madrid in spreading the virus. Madrid was the main local outbreak at the onset of the pandemic. Later, local peaks of incidence and mortality appeared in other provinces with a high level of mobility from and to Madrid in the week before the onset of the local outbreaks (Mazzoli et al., 2020).

Table 6.2: Epidemiological variables per 100,000 people in Spain's regions (May 29^{th}, 2020).

Region*	Cases	Deaths
Andalucía (Andalusia)	209	17
Aragón	537	68
Asturias [Principado de ...]	328	32
Canarias (Canary Island)	110	7
Cantabria	495	36
Castilla y León	1,077	84
Castilla-La Mancha	1,190	147
Cataluña (Catalonia)	791	96
Ceuta	210	5
Comunitat Valenciana	281	29
Extremadura	327	48
Galicia	422	23
Islas Baleares (Balearic Islands)	185	20
La Rioja	1,713	114
Madrid [Comunidad de ...]	1,086	137
Melilla	155	2
Murcia [Región de ...]	175	10
Navarra (Navarre)	1,277	79
País Vasco (Basque Country)	832	66

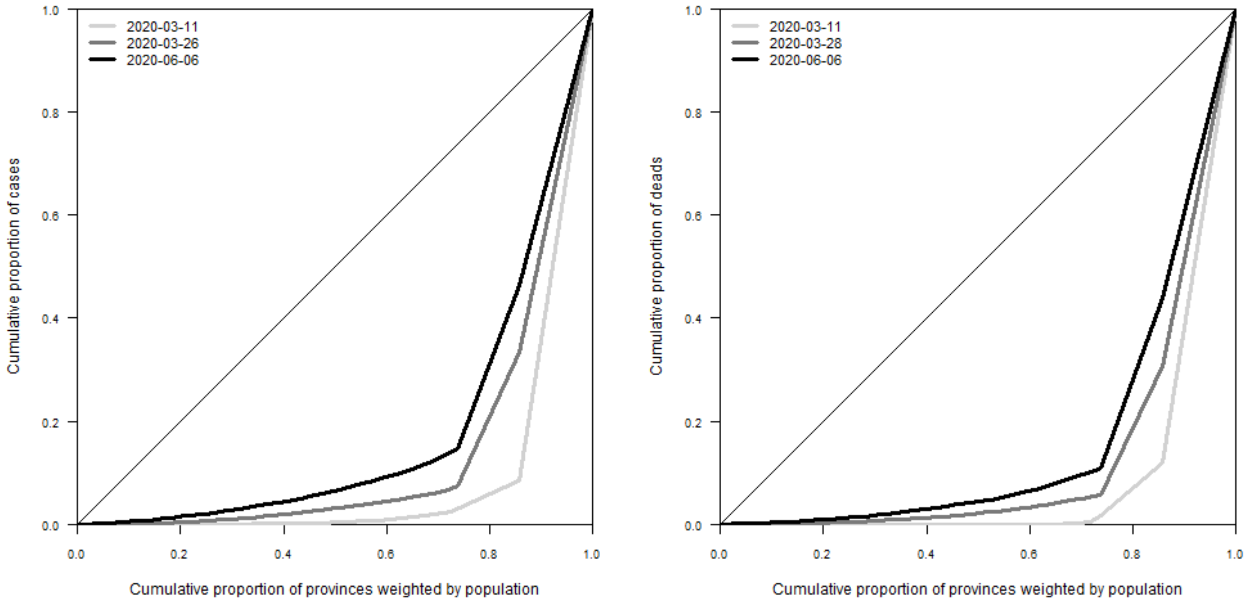
* In brackets the full name and in parentheses the English name of the region (where applicable)

The data are completed with Table 6.2, which quantifies the cumulative cases and deaths per 100,000 head of population at the end of the outbreak. It can be seen that the regions with most cases are those closest to the initial outbreaks: Comunidad de Madrid and its neighboring regions, Castilla-La Mancha and Castilla y León, plus La Rioja and its neighbor Navarra. With respect to the deaths, the three regions with ratios above 100 are Castilla-La Mancha, Comunidad de Madrid and La Rioja. These figure contrasts with the lower ratios of less than 25 in Andalusia and Galicia (apart from the islands and autonomous cities).

In summary, descriptive data shows that as time went, cases and deaths became more evenly distributed across provinces, though the distribution at the end of the outbreak remained quite heterogeneous. Inequality indices can thus be expected to show a decreasing trend over the outbreak. In the next subsection, these indices are quantified.

6.5.2 Inequality measures

Figure 6.5 compares the Lorenz curves for cumulative cases and deaths on key dates. These comparisons show that provinces in the low and middle parts of the distribution increase their share of the total cumulative



(a) Cumulative cases

(b) Cumulative deaths

Figure 6.5: Lorenz Curves for the epidemiological variables on key dates.

cases and deaths over time, bringing the curve closer to the diagonal though still far from it. Thus, it can be unambiguously claimed that the distributions of cumulative cases and deaths became more homogeneous as the Covid-19 disease evolved until the first wave ended.

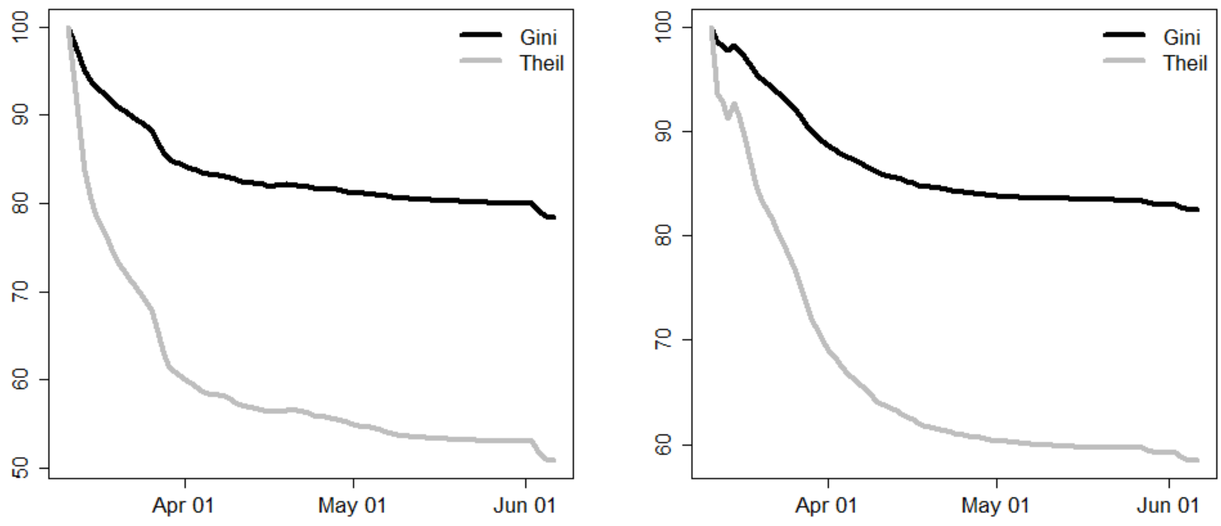
Table 6.3: Epidemiological variables inequality across provinces.

	Cumulative Cases		Cumulative Deaths	
Day	Gini	Theil*	Gini	Theil*
Onset	0.83	1.60 (0.41)	0.82	1.58 (0.40)
Peak day	0.69	0.93 (0.24)	0.69	0.96 (0.24)
Ending	0.65	0.81 (0.21)	0.68	0.92 (0.23)

* In parenthesis the value normalized by the maximum level of the Theil index ($Ln(52) = 3.95$)

Along with this graphic result, inequality in the provincial distribution of the epidemiological variables is quantified using the Gini and Theil indices defined in expressions (6.1) and (6.2), respectively. Table 6.3 shows the indices for cumulative cases and deaths on key dates. Given that the Theil index is unbounded above, it is normalized by the maximum level that it can reach ($Ln(M)$ with M being the number of provinces); this normalisation can be called the Relative Theil Index (Bellù and Liberati, 2006). It enables the Gini and Theil indices to be compared since both ranged from 0 to 1.

The results in Table 6.3 highlight two important facts. First, at the end of the Covid-19 wave, it can be stated that the numbers of cumulative cases and deaths are fairly unevenly distributed among the Spain's provinces. The Gini index shows similar inequality for cases and deaths, ranging at the ending of the wage from 0.65 to 0.68. It is worth noting that this figure is twice the Gini index for the distribution of disposable income among the Spanish population which was 0.33 in 2018 (Eurostat, EU, 2020a). Second, as the Lorenz curves show in Figure 6.5, the level of inequality quantified for both indices decreases over time as the virus spreads.



(a) Cumulative cases

(b) Cumulative deaths

Figure 6.6: Trend in inequality for the epidemiological variables. From provincial distributions with population weight.

This last result is illustrated in more detail in Figure 6.6 where the trend in the indices for the full period studied (March 11 to June 6, 2020) is displayed. The trend is shown with both indices normalised to 100 for the onset day. Positive and negative fluctuations in inequality place indices above and below 100, respectively. This enables the scale of the changes in the Gini and Theil indices to be easily compared in time and with each other.

As expected, Figure 6.6 shows a monotonic downward trend in the inequality in both cumulative cases and deaths. The only exception appears for March 15 when inequality in deaths increases slightly, probably because the daily deaths reported were abnormally low. It can be observed that inequality in the distribution of cases decreased by between 22% and 49% from the onset to the ending, depending on the index. Likewise, inequality in the distribution of deaths decreased by between 17% and 42% in the same period.

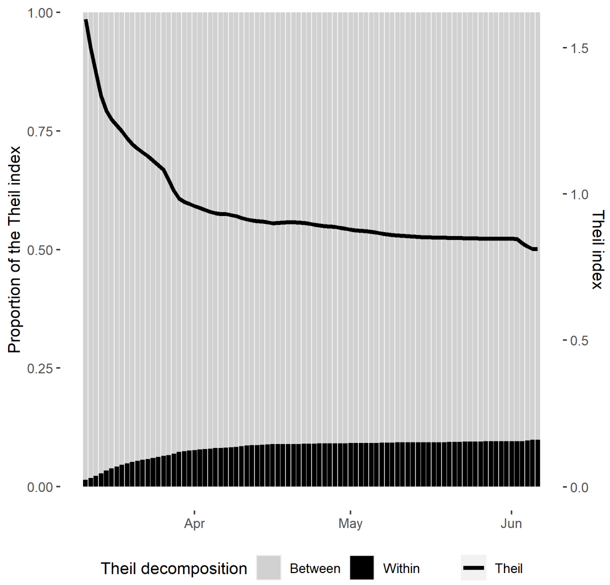
6.5.3 Decomposition of inequality by regions

Although the trend is monotonically decreasing, two different periods are observed. In the first part of the wave, in March, inequality decreases very fast as the disease spreads from the initial outbreaks, located mainly in Madrid, to the rest of the country, reaching the surrounding provinces very rapidly. As pointed out by Mazzoli et al. (2020), the emergence of local peaks of incidence and mortality was closely correlated with mobility from and to Madrid in the early-stage weeks. This spread leads to accumulate new cases and deaths more homogeneously across provinces. From mid-April onwards, inequality decreases at very low rates, until a stationary level is reached. Note that a stationary inequality level does not necessarily mean that the number of cases or deaths is stationary, but that those which occur are distributed maintaining the same unevenness across the provinces. In fact the number of cases and deaths falls significantly from mid-April to late May (see

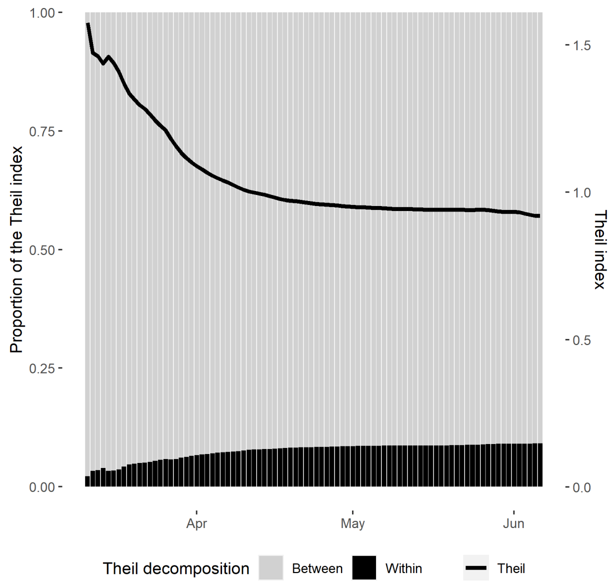
Figure 6.3).

Another noteworthy feature illustrated in Table 6.3 and Figure 6.6 is that Gini and Theil indices show similar trends, but the differences between them are significant in quantitative terms. The Theil index shows a greater variation in inequality over time, because it is more sensitive to changes in the tails of the distribution, while the Gini index is more sensitive to changes in the middle of the distribution than at the top and bottom (Allison, 1978; Atkinson, 1970). In our case, the spread of the virus affected provinces at the tails, especially in March. In those days new cases and deaths emerged in Madrid (upper tail), while many provinces had hardly any (low tail). This explains why the Theil index decreased more sharply than the Gini index in the first part of the wave.

The decomposability property of the Theil index enables it to be calculated how much of the inequality observed in the epidemiological variables can be explained by differences *between* regions and how much by differences across provinces *within* those regions. Given that in Spain authority for planning and management of health services is devolved to regional authorities, the *between* component may reflects differences between regional health services (investment, human resources, governance, etc.) among other things. The *within* component represents the differences across provinces *within* each region. These differences may reflect idiosyncratic characteristics of provinces that belong to the same region (population density, aging population, big cities, airports, etc.).



(a) Cumulative cases



(b) Cumulative deaths

Figure 6.7: *Between* and *within* regions inequality decomposition of the Theil index.

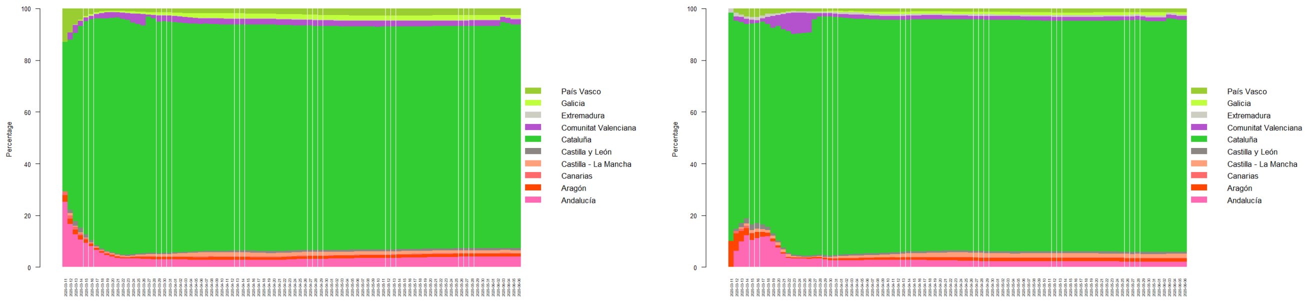
The *between* and the *within* inequality components of the Theil index for epidemiological variables in Spain are calculated according to expressions (6.3)-(6.5). Figure 6.7 shows their trend for the full period studied (March 11 to June 6, 2020). Table 6.4 supplements this information by showing the contribution of each component in quantitative terms on the key days.

The most striking finding is that inequality in both epidemiological variables can be attributed mostly to differences between regions in the period analysed. Indeed, there are hardly any differences in the percentage

Table 6.4: *Between* and *within* decomposition of the epidemiological variables inequality across regions.

	Cumulative Cases		Cumulative Deaths	
Day	*Between*	*Within*	*Between*	*Within*
Onset	98.57%	1.43%	97.76%	2.24%
Peak day	93.47%	6.53%	94.13%	5.87%
Ending	90.11%	9.89%	90.84%	9.16%

contribution of each component to total inequality when cases and deaths are compared. Furthermore, the gap between the *between* and *within* components widens over time. The *between* component can be asserted to account for over 90% of the inequality while the *within* accounted for less than 10% while the outbreak was active.



(a) Cumulative cases

(b) Cumulative deaths

Figure 6.8: Contributions of regions to the within inequality distribution for the epidemiological variables.

The Theil decomposition also enables the contribution of each region to the *within* inequality to be computed using expression (6.4). In this case the *within* component accounts for just a small part of the total inequality, but it is still useful to analyse the role played by each region in the contribution to this component. Figure 6.8 shows the trend in the contributions by the regions to the *within* component throughout the wave for cumulative cases and deaths. Notice that single-province regions do not appear in Figure 6.8 because intra-provincial inequality is meaningless when there is only one province. The most significant finding shown in Figure 6.8 is that Catalonia accounts for over 86% of the *within* inequality at the end of the wave. This occurs for both epidemiological variables and persists throughout most of the wave.

The expression that defines the *within* component of inequality, (6.4), reveals that two elements determine the contribution of each region to that component. The first is the proportion of the epidemiological variables of each region in terms of that variable for the whole country weighted at provincial level ($\sum_{i=1}^{M_r} p_{i,r} \cdot e_{i,r} / \sum_{i=1}^{M} p_i e_i$). The second is the inequality of the distribution of the variable within each region (T_r). Both items are shown to be multiplying in expression (6.4). This means that the contribution of a particular region is high only when both components are high, as in the case for Catalonia. On the one hand, cumulative cases and deaths in Catalonia are over-represented given the populations of its provinces. At the end of the wave, cumulative cases and deaths in Catalonia counted for around 33% of the total when weighted at the provincial level, while its population represents 16% of the Spanish population. On the other hand, the four provinces of Catalonia were hit very unevenly by COVID-19. Cumulative deaths in the province of Barcelona at the end of the wave were 11 percentage points higher than those corresponding to its population, while those of Tarragona were 6 percentage points lower. This means there was high inequality within Catalonia. These two elements explain the high contribution of Catalonia to the *within* inequality.

Table 6.5: Contributions of Spain's regions to the *within* inequality in the epidemiological variables at the end of the wave (June 6^{th}, 2020).

	Cumulative Cases				Cumulative Deaths			
	T_{within}				T_{within}			
Regions*	Value	Share	T_r	Weights**	Value	Share	T_r	Weights**
Andalucía	0.00323	4.02%	0.1309	2.47%	0,00178	2.11%	0.1093	1.63%
Aragón	0.00088	1.09%	0.1446	0.61%	0.00107	1.27%	0.1650	0.65%
Canarias	0.00026	0.32%	0.0914	0.28%	0.00017	0.20%	0.1092	0.16%
Castilla-La Mancha	0.00088	1.10%	0.0714	1.23%	0.00127	1.50%	0.0959	1.32%
Castilla y León	0.00063	0.78%	0.0697	0.90%	0.00068	0.81%	0.1096	0.63%
Cataluña	0.06947	86.50%	0.2141	32.44%	0.07564	89.71%	0.2245	33.69%
Comunitat Valenciana	0.00166	2.07%	0.0521	3.19%	0.00123	1.46%	0.0458	2.69%
Extremadura	0.00015	0.19%	0.0812	0.19%	0.00053	0.63%	0.2518	0.21%
Galicia	0.00108	1.34%	0.0987	1.09%	0.00071	0.84%	0.1437	0.49%
País Vasco	0.00209	2.60%	0.1182	1.76%	0.00123	1.46%	0.1055	1.16%
TOTAL	0.08031				0.08431			

* Single-province regions do not appear because they do not contribute to the within inequality.

** $\left(\sum_{i=1}^{M_r} p_{i.r} \cdot e_{i,r}\right)/\left(\sum_{i=1}^{M} p_i \cdot e_i\right)$ where e represents cases or deaths.

Other regions also show high levels of intra-regional inequality. This is the case of Extremadura and Aragón for cumulative deaths and Aragón and Andalucía for cumulative cases. However, none of these regions has numbers of cumulative cases or deaths that are over represented given their populations at provincial level. This is shown in Table 6.5, which summarizes the contribution of each region to the *within* component at the end of the wave (June 6, 2020) distinguishing between the two elements involved.

6.6 Discussion and conclusions

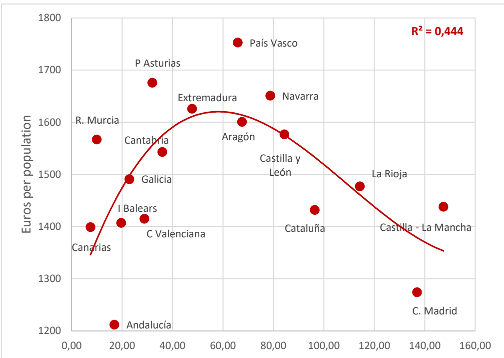
The COVID-19 pandemic is leaving a huge number of infected persons and deaths throughout the world. However, the disease has not spread homogeneously across or within countries. Pending the arrival of data on the effects of the virus worldwide, this study analyses the distribution of epidemiological variables in Spain, a country where the first wave can be considered to have developed from early March 2020 to early June 2020.

An awareness of the existence and the nature of inequalities observed in epidemiological variables is necessary to develop successful policies for improving and homogenizing the planning and management of health services in future waves. This issue is especially relevant in Spain, where the health system is decentralized, with regions responsible for health care planning, organization and management and thus are politically accountable to their constituents (Costa-Font and Moscone, 2009).

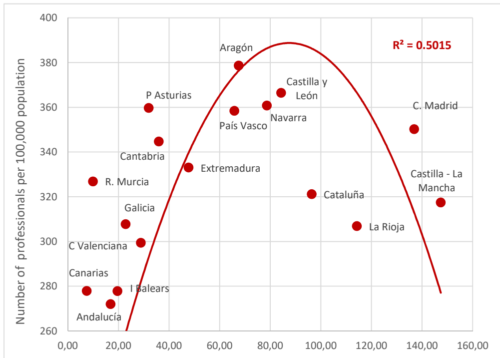
This paper analyses epidemiological variables during first wave of COVID in Spain to assess how evenly they were distributed throughout the provinces of Spain. Using standard inequality metrics, the study shows that by the end of the first wave cumulative cases and deaths were fairly unevenly distributed across Spain's provinces, with a level of inequality twice that observed for the distribution of disposable income across the

Spanish population during the last ten years. Moreover, the study also shows that over 90% of the inequality observed in COVID epidemiological variables can be attributed to differences between regions.

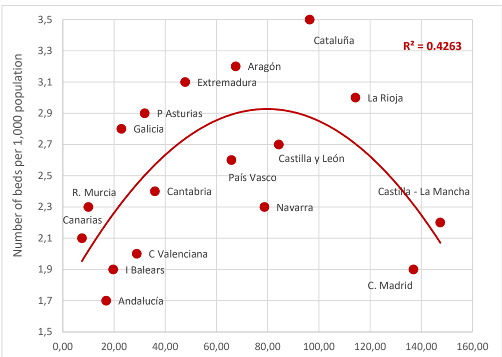
It is worth noting that our analysis is straitened to the quantification of the unevenness. The lack of data prevents a more in-depth statistical analysis that allows establishing, in terms of causality, which factors behind the unevenness observed in the distribution of the epidemiological variables across regions and provinces. Technically, once the wave is over, there is only one observation on the distribution of the variables of interest in the territory; this makes it impossible to establish causality between the distribution of these variables and the potential determining factors. Nevertheless, descriptive statistical analysis of the Spanish regions allows to discern what factors that may have led to an uneven distribution of cases and deaths.



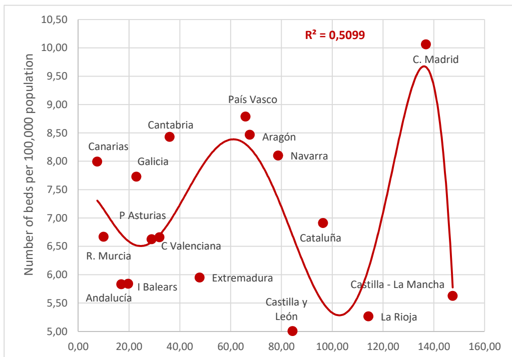
(a) Public health expenditures financed by regions, 2018.



(b) Healthcare professionals hired by the public system, 2018.



(c) Hospital Beds in the public system, 2018.

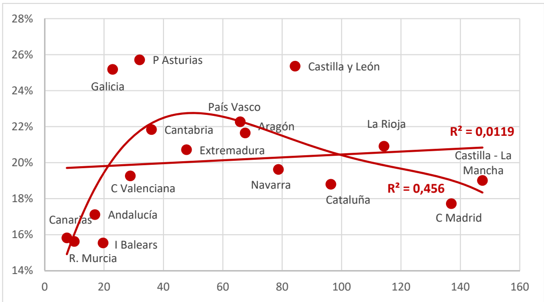


(d) Intensive Care Beds, 2011.

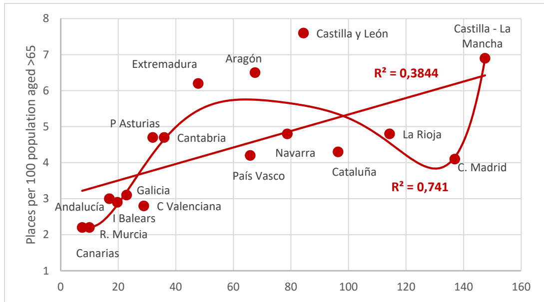
Figure 6.9: Health effort vs. cumulative deaths. X-axis: COVID-19 Deaths per 100,000 people. Red line: polynomial adjustment. Data source: (a) and (b) Spanish Health Ministry (Rodríguez Blas, 2020), (c) Spanish Health Ministry (2020) and (d) Martín et al. (2013).

The fact that most of the inequality observed in COVID epidemiological variables can be attributed to differences between regions could be seen as due, among other things, to the response of the regional health authorities to the pandemic being very diverse and ending up generating differentiated effects. There is still no data on these responses at regional level. However, data prior to the outbreak glimpse great differences in the management of the health system at regional level that may have affected incidence unevenly. Figure 6.9

shows the empirical relation between the COVID-19 deaths and some indicators that measures the health effort made by the regions. We see that there is not a clear negative relationship between health effort and deaths. Indicators such as the per capita public health expenditure financed with the regions' own funds (panel 6.9(a)), the number of healthcare professionals hired by the public system (panel 6.9(b)) or the number of hospital beds available (panel 6.9(c)) show a bell shape when they are crossed with the number of deaths. Only if we focus on the regions hardest hit by the virus, we see that there is a clear negative relationship between health effort and the number of deaths. It seems that the regional health systems are prepared to serve a certain number of patients. In those regions where the virus was particularly virulent, health capacity was saturated and, given this limitation, regions with fewer resources suffered a higher incidence of deaths. Notice however that the relationship between deaths and the number of intensive care beds is more complex (panel 6.9(d)), although it must be taken into account that this indicator has not been officially collected since 2002 and the data used come from a study carried out by means of a questionnaire answered by hospitals (Martín et al., 2013).



(a) Population aged 65 and over, 2019.

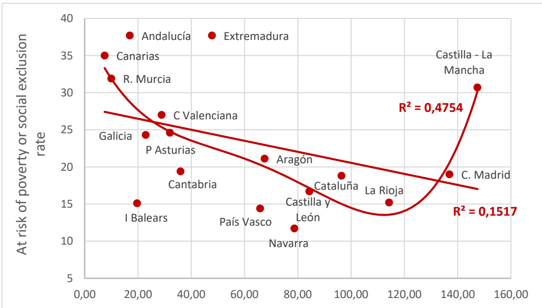


(b) Nursing home places, 2019.

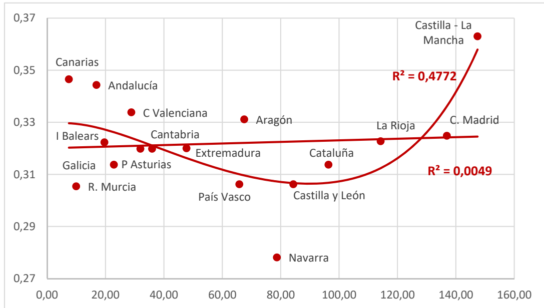
Figure 6.10: Aging factors vs cumulative deaths. X-axis: COVID-19 Deaths per 100,000 people. Red lines: lineal and polynomial adjustment. Data source: (a) INE (2019a) and (b) Abellán García et al. (2019).

There are many other factors that may have led to an uneven distribution of cases and deaths. Recent studies have shown that people with some health and socioeconomic personal characteristics are more likely to develop a severe form of Covid-19. In particular, age has been confirmed as a critical factor related to COVID-19 deaths (Moosa and Khatatbeh, 2020, in press; Williamson et al., 2020). There are still no official data on deaths from the disease per age group in Spain. Figure 6.10(a) illustrates the positive correlation between the deaths and the proportion of population aged 65 years and more for the Spanish regions. The correlation is weak over the whole sample (0.34), although more relevant between the regions less affected by the virus (0.85 for regions with less than 40 deaths per 100,000 people). In fact, regions such as Galicia or Asturias with over 25% of their population classed as elderly show death numbers under 40 per 100,000 people. However, this positive relation is not observed at this aggregate level with other medical risk factors as obesity or diabetes which have been also shown as determinants in progressing to severe forms of COVID (Cai et al., 2020; Guo et al., 2020; Simonnet et al., 2020; Williamson et al., 2020).

Moreover, the COVID-19 pandemic revealed the precarious position of nursing homes in Spain (Rada, 2020). Most nursing homes do not have doctors or nurses on staff. During the peak days of the pandemic, nursing homes in Madrid received guidelines from the Health Department indicating that residents with respiratory infection symptoms should not be sent to hospital. There are no official records for mortality in nursing homes, but Comas-Herrera and Zalakain (2020) estimate that between March 8 and April 8 there may have been 9,756 deaths, which would account for 57% of the total deaths due to COVID-19 in Spain to that date. A recent study by Abellán García et al. (2019) has established the number of places in Spanish nursing homes at regional level. Castilla-La Mancha, the most affected region by the virus in terms of cases and deaths per population, is the region with the second highest ratio of places by population over 65. Crossing these data with those of deaths from Covid-19 during the first wave, a moderate linear correlation of 0.67 is observed (see Figure 6.10(b)).



(a) AROPE indicator, 2019.

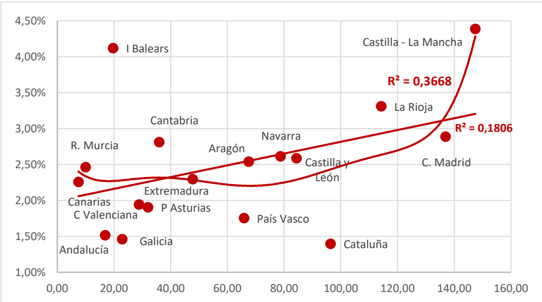


(b) Gini coefficient for regional income, 2012.

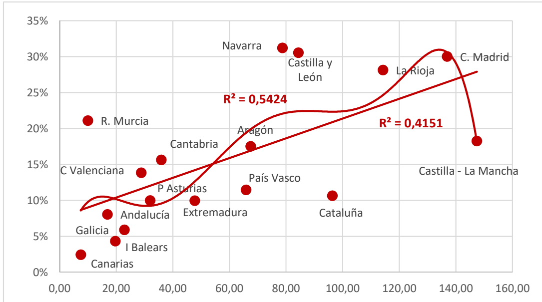
Figure 6.11: Deprivation and income inequality vs. cumulative deaths. X−axis: COVID−19 Deaths per 100,000 people. Red lines: lineal and polynomial adjustment. Data source: (a) INE (2019a) and (b) Jurado Málaga and Perez-Mayo (2014).

Additionally, the diffusion of the virus does not only depend on medical individual characteristics, but also on their socio-economics situation (Williamson et al., 2020). In this context, the Spanish regions also show large disparities. The AROPE indicator, which measures the percentage of population at risk of poverty or social exclusion (Eurostat, EU, 2012), shows substantial differences between the Spanish regions. There is a difference of 26 percentage points between the best and worst positioned regions (Navarra, 12% and Andalucía, 38%, respectively). These disparities, however, do not show a positive correlation with deaths caused by COVID at the regional level. On the contrary, when both variables are crossed, a negative correlation emerges from the whole sample (Pearson coefficient -0.39). The correlation becomes strongly positive when the sample is limited to the regions more affected by the virus (0.78 for regions with more than 70 deaths per 100,000 people), which confirms the highly non-linear relationship shown by the data (see Figure 6.11(a)). A similar U-type relationship is seen when deaths are crossed with the regional Gini index that measures how unevenly income is distributed within each region (see Figure 6.11(b)).

Another factor that may have led to an uneven distribution of cases and deaths was the mobility (Henríquez et al., 2020; Siqueira et al., 2020). Some studies point out that the virus arose mainly in Madrid and spread rapidly to the closest provinces during the early-stage weeks because of mobility from and to Madrid (Mazzoli et al., 2020). In this sense, regions furthest from Madrid, such as Galicia, Murcia and the islands were less affected by the disease in the onset of the outbreak. Figure 6.12 shows the empirical regional relationship between



(a) Population moved inter-regionally, 2019.



(b) Undergraduate students residing in other regions, 2018-19.

Figure 6.12: Mobility indicators vs. cumulative deaths. X−axis: COVID−19 Deaths per 100,000 people. Red lines: lineal and polynomial adjustment. Data source: (a) INE (2019b) and (b) Ministerio de Universidades (2019).

deaths from COVID and two variables that can be considered mobility proxies: percentage of the population that has moved inter-regionally during the last year (Figure 6.12(a)) and percentage of the undergraduate students residing in other regions (Figure 6.12(b)). Both variables can indicate the mobility of a part of the population that frequently moves from their work places to their homely places. This type of movement was very generalized during the days before the State of the Alarm in which schools closed and teleworking was encouraged by many companies. We see that both variables shows a positive correlation with the deaths at regional level (0,43 and 0.64, respectively), which allows us to surmise that mobility plays a relevant role in the distribution of deaths.

Figure 6.12 also shows remarkably low mobility levels for Catalonia, especially when compared to other regions with a similar population and economic level. Notice, for instance, that inter-regional mobility in Catalonia is less than 1.5% (the lowest level among all regions) vs almost 3% in Madrid; in the same sense, 10% of undergraduate students in Catalonia come from other regions vs more than 30% of those in Madrid. These figures probably reflect that issues such as the widespread use of Catalan as the spoken language or the turbulent political moment experienced after the unilateral declaration of the Catalan Republic in 2017, may be discouraging Catalonia inter-regional mobility both from and to other regions. However, mobility in Catalonia is higher when inter-provincial mobility is considered. In fact, more than 7% of the population of each of the four provinces in Catalonia moved their residence to or from other provinces in the last year, which is substantially higher than the 3% shown by Madrid (which is a uni-provincial region). These high levels of the inter-provincial mobility in Catalonia may be one of the causes behind the high results obtained for inequality within Catalonia (see Figure 6.8).

Notwithstanding all the empirical evidence just mentioned, more comprehensive studies should be carried out to determine the underlying causes of the uneven distribution of the COVID epidemiological variables observed. Likewise, there is a perceived need to analyze the distribution of the impact of COVID worldwide, taking into account the subsequent waves that are taking place. At the present time, he first half of January 2021, the disease seems to be far from being controlled mainly in Europe and America. An analysis of the distribution of epidemiological variables across countries will complete the picture in a more understandable way.

Finally, we must not lose sight that an accurate metrics on how the prevalence of COVID-19 is distributed territoriality may enable good practices developed by regions against the epidemic to be identified, so that more efficient responses can be provided in current and future outbreaks.

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Appendix 6.A Spain composition

Table 6.A.1: Spanish regions and provinces.

Region*	Provinces	Main data source
Andalucía (Andalusia)	8: Almería, Cádiz, Códoba, Granada, Huelva, Jaén, Málaga, Sevilla	Junta de Andalucía (press release)
Aragón	3: Huesca, Zaragoza y Teruel	Gobierno de Aragón (website)
Canarias (Canary Island)	2: Las Palmas y Santa Cruz de Tenerife	Consejería de Sanidad del Gobierno de Canarias (press release)
Cantabria	1: Cantabria	Instituto de Salud Carlos III
Castilla y León	9: Avila, Burgos, León, Palencia, Salamanca, Segovia, Soria, Valladolid, Zamora	Junta de Castilla y León (website)
Castilla-La Mancha	5: Albacete, Ciudad Real, Cuenca, Guadalajara, Toledo	Gobierno de Castilla-La Mancha (press release)
Cataluña (Catalonia)	4: Barcelona, Girona, Lleida, Tarragona	Generalitat de Cataluña and Portal de Transparència Catalunya
Ceuta	1: Ceuta	Instituto de Salud Carlos III
Comunidad de Madrid	1: Madrid	Instituto de Salud Carlos III
Comunitat Valenciana	3: Alicante / Alacant, Castellón / Castelló, Valencia / València	Generalitat Valenciana (website)
Extremadura	2: Badajoz, Caceres	Junta de Extremadura
Galicia	4: A Coruña, Lugo, Orense, Pontevedra	Sergas: Servicio Gallego de s Salud (press release)
Islas Baleares (Balearic Islands)	1: Baleares	Instituto de Salud Carlos III
La Rioja	1: La Rioja	Instituto de Salud Carlos III
Melilla	1: Melilla	Instituto de Salud Carlos III
Navarra	1: Navarra	Instituto de Salud Carlos III
País Vasco (Basque County)	3: Araba/Álava, Bizkaia, Gipuzkoa	Gobierno Vasco. Osakidetza (dashboard, press release)
Principado de Asturias	1: Asturias	Instituto de Salud Carlos III
Región de Murcia	1: Murcia	Instituto de Salud Carlos III

* In parentheses the English name of the region (where applicable)

Conclusions

Their intensive exploitation as well as the climate variations are adversely affecting the future of some natural resources (UN, 2020a,b). Global initiatives have been set in the last decades in order to prevent their exhaustion and encourage an equitable distribution across countries (UN, 2015; CBD, 2010). In this respect, the Blue Economy is being promoted as a path to obtain the largest benefits offered by oceans while ensuring its recovery and future sustainability. Multiple countries are already committed to Blue Growth Strategies at a global level. These strategies aim to maximize the revenues sustainably obtained from the ocean based on a combination of the economic, ecological, social and scientific ocean knowledge. In particular, these strategies empower the development of new ocean activities such as ocean tourism, transport, energy and seabed mining (High Level Panel for a Sustainable Ocean Economy, 2020; EC, 2017, 2012). Whereas specific fishing industry improvements are proposed to increase the efficiency of this activity and minimize the damage to the environment in some Blue Growth Strategies (High Level Panel for a Sustainable Ocean Economy, 2020), the fishing industry seems underestimated in other ones (Saviolidis et al., 2020; Cisneros-Montemayor et al., 2019; Boonstra et al., 2018). Giving suddenly the central role to new activities obviates the large social and economic value represented by the fishing industry for many communities as well as its global contribution to human health (Cisneros-Montemayor et al., 2019).

Urgent global policies are required for the recovery of the over-exploited fish stocks, which represent over one third of their total (FAO, 2020). Achieving good conditions of the fish stocks is necessary for the whole environment and the prosperity of ocean economic activities (UN, 2015). Even though the conditions of fish stocks may be determined by multiple causes, the fishing activity and the environmental changes have a significant impact (FAO, 2020; Perissi et al., 2017; Lotze and Worm, 2009). Therefore, reducing the pressure exerted by the fishing industry on the over-exploited marine ecosystems could facilitate their recovery by increasing their capacity to replenish (Perissi et al., 2017). Similarly, mitigating the environmental impact of human activities could avoid magnifying the adverse effects of the climate change experienced in fisheries (IPCC, 2019). The present thesis examines both of these aspects by focusing on multiple aspects related to the sustainable exploitation of fisheries such as the resources distribution and the production of discards along the first four chapters and on the minimization of the environmental impact derived from human food consumption in the fifth chapter.

From Chapter 1 to Chapter 3, the distribution of fishery resources is analyzed. Besides the normative values attributed to equity in the distribution of resources (Bennett, 2018), it is recognized as a motor for the expansion of the global economic growth (Akinci, 2018; Berg et al., 2018; IMF, 2017). Furthermore, the perception of unevenness in the distribution of resources by fisheries stakeholders has been found to difficult their cooperation for the sustainable management of particular fish stocks (Kourantidou et al., 2021; Matić-Skoko and Stagličić, 2020; Forse et al., 2019; O'Higgins and O'Hagan, 2019; Napier, 2016; Fabinyi et al., 2015). Therefore, equity in the distribution of fishery resources has been established as one of the pillars to achieve Blue Growth (High Level Panel for a Sustainable Ocean Economy, 2020). Chapter 1 analyzes the distribution of common global resources across countries in the last decades in order to provide a broad perspective on the responsibility of fisheries exploitation for the design of effective and socially accepted global policies targeting the conservation of fish stocks. Even though the distribution of common global fish resources seems to have become more homogeneous from 1961 to 2014, this chapter shows that significant differences still exist across countries in terms of catches. While differences in the catches of countries were mainly determined by technological dissimilarities in the beginning of the period analyzed, catches heterogeneity seems mostly motivated by differences in the harvesting area at the end of the period analyzed. The decrease of the technological relevance on catches heterogeneity could be explained by the technological advances occurred in the fishing industry during these decades (Valdemarsen, 2001) whereas the increase of the harvesting area relevance could be motivated by their previous exploitation patterns and their adaption to climate change (FAO, 2020). The EU is an example of the relevance of international fishery resources distribution. Since the 1970s, EU MSs are committed to a common regulation for the sustainable exploitation of their fisheries (EC, 2019, 2013). Despite the regulation has been adapted over time to match with ecological and socioeconomic requirements (EC, 2019), its continuance is threatened by the slow recovery of fish stocks and the social tensions arisen with the heterogeneous distribution of fishery resources (Cardinale et al., 2017; Garza-Gil et al., 2017; Orach et al., 2017; Da-Rocha et al., 2012). Aiming to contribute to the homogenization of the EU fishery resources distribution and the adaption of its fishing activity to more sustainable exploitation pattern, Chapter 2 deepens in the technological reasons producing the catches inequality observed across EU MSs from 2008 to 2016. During this period, the catches of EU MSs seem to have become more alike. However, significant differences remain across countries operating within each of the two major fishing areas (the Atlantic waters, ATW, and the Mediterranean and Black Sea, MBS). Within the ATW, the fishing labor and the technical progress are the main reasons for the catches inequality observed whereas the technological productivity and the capital intensity contribute to homogenize catches distribution. Within MBS, fishing labor and technological productivity are the largest contributors to catches inequality followed by the technical progress and the capital intensity. Therefore, policies seeking a sustainable exploitation of each fishing area as well as an even distribution of resources should adjust the fleets of countries towards those of the most sustainable ones incising specially in the factors generating the largest heterogeneity in each area (fishing labor and technical progress in the ATW and fishing labor and technological productivity in the MBS). Nevertheless, especial attention should be placed when modifying the fishing labor as it may involve notorious socioeconomic consequences. Distribution of fishery resources within countries has also significant social, economic and environmental consequences (Fabinyi et al., 2015; Garza-Gil and Varela-Lafuente, 2015). Chapter 3 analyzes the distribution of catches across vessel owners operating in the Norwegian fisheries. The analysis focuses on the concentration of catches across vessel owners within the major fishing groups in an attempt to reflect the quota trade restrictions across different vessel groups (Hannesson, 2013). After looking at the evolution of the concentration in each case, the determinant factors are analyzed. As pointed out in previous literature, concentration seems to have increased in all fishing groups. Even though the reduction in the number of owners played a significant role in the increase of catches concentration, variations in differences of the remaining owners had the largest impact on the increase in the concentration of catches within most of the fishing groups. These results confirm an structural change in the Norwegian fish supply, which is experiencing a concentration of catches in fewer owners. Being aware of the factors rising the concentration in the fish supply allows to design policies preventing an excessive concentration of market power.

In Chapter 4, the factors underlying the decision to discard catches in the fishing activity are examined. Discarded catches are associated to private, social and environmental costs. From a private perspective, costs arise by the use of input factors without later revenue (Clucas, 1997). From the social perspective, discarding catches imply wasting edible resources that could help to ensure food security (FAO, 2020). From the environmental perspective, discards alter the balance of the ecosystems of origin and decomposition (Clucas, 1997). Even though there is not a global regulation addressing discarded catches, several regions have implemented unilaterally different types of policies to prevent discarding in the fishing activity (Condie et al., 2014). Chapter 4 analyzes which have been the factors motivating the catches discarded by countries in the last decades aiming to contribute to the design of efficient global policies minimizing discards. In line with previous literature, estimations from this chapter highlight the need to improve the selectivity of certain fishing gears (Herrmann et al., 2020; Tang et al., 2019; Vogel et al., 2017) and incentive fish demand (Van Putten et al., 2019) to prevent fishers from discarding catches. In addition, results from this chapter suggest the re-orientation of the activity of fishing gears towards species for which they are more efficient.

In Chapter 5, the implementation of fiscal policies for the reduction of the environmental impact of food consumption is evaluated. Food consumption patterns are raising concerns given the environmental consequences derived from the production of the raw ingredients to the final household consumption (FAO et al., 2021). Among the environmental consequences of food consumption are the generation of greenhouse gas emissions (Aguilera et al., 2021), the use of water (Blas et al., 2019) and the loss of edible resources (Gustavsson et al., 2011). Chapter 5 studies the optimal fiscal policies to be implemented in the case of Spain to minimize each of these environmental impacts of the food consumption as well as the combination of all of them. This chapter highlights the increase in the efficacy of policies derived from considering the whole food consumption rather than just focusing on the most harmful categories for the environmental impact of interest as the relationships between them facilitate the direction of consumption towards more environmentally friendly food. Clearly defining the environmental target is of utmost importance as optimal fiscal policies are contradictory for some environmental impacts. In the case of fish consumption, subsidies should be applied to minimize the greenhouse gas emissions and the use of water whereas taxes should be charged to minimize food loss and waste. Given the relationship of substitution observed between meat and fish, subsidies to the latter allow to decrease the consumption of the former, which constitutes one of the main food categories generating greenhouse gas emissions and requiring water. The relationship of substitution also found between products in the remaining food category and fish shows that the subsidies to fish consumption also imply a reduction of consumption from the remaining food category, which is the food category with the largest water requirement. By contrast, the complementary relationship between fruit and vegetables and fish allows to decrease the consumption of the former when implementing taxes on the latter for the minimization of the food loss and waste. Being contradictory across the environmental aspects studied, fiscal policies are less effective when their combination is targeted. Setting higher fiscal limits within socially acceptable levels is associated with further reductions of the environmental impacts. Nevertheless, their socioeconomic and health implications should not be disregarded.

Examining the previous subjects allows to approach some of the multiple dimensions affecting fisheries sustainability from a socioeconomic perspective. Considering these socioeconomic matters in fisheries management allows to contextualize the fishing activity, designing regulatory frameworks that benefit the environment as well as the whole society (High Level Panel for a Sustainable Ocean Economy, 2020).

During the elaboration of the present thesis arose COVID-19 pandemic (WHO, 2020), having important socioeconomic consequences at a global level (Ferreira et al., 2021). The especial circumstances set by the national State of Alarm and the experience on inequality analysis acquired in Chapters 1 and 2 lead to the implementation of these techniques to improve the understanding of the ongoing situation. Chapter 6 analyzes the differences in the effects of this pandemic during its first wave in Spain across Autonomous Communities. Even though the incidence and deaths were located in a few Communities at the beginning, they spread across the whole country over time. Having decreased over time, differences across Communities remained until the first wave ended, reflecting the need to homogenize health services within the country using as reference those Communities with the most efficient handling of the pandemic.

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