THE UNIVERSITY OF BRITISH COLUMBIA

Institute for the Oceans and Fisheries

WORKING PAPER SERIES

Working Paper #2022 - 04

A rich analysis of the economic, social and environmental effects of harmful fisheries at the ecosystem level

Sumaila, U.R., Zeng, Z., Lam, V.W.Y, and Cheung, W.W.L

Year: 2022
Email: r.sumaila@oceans.ubc.ca

This working paper is made available by the Institute for the Oceans and Fisheries, University of British Columbia, 2202 Main Mall, Vancouver, BC, V6T 1Z4, Canada

A rich analysis of the economic, social and environmental effects of harmful fisheries at the ecosystem level

Sumaila, U.R.,[1,2] Zeng, Z. [1], Lam, V.W.Y[1], and Cheung, W.W.L[1]

1. Institute for Oceans and Fisheries, University of British Columbia
2. School of Public Policy and Global Affairs, University of British Columbia

Abstract
Marine fish stocks and the fisheries they support play a vital role in meeting the food and nutritional needs of tens of millions of people worldwide while providing jobs and incomes to many more millions. Even though marine fish stocks are renewable they are seriously threatened by many stressors, including overfishing due to ineffective management and the perverse effects of government policies such as the provision of harmful subsidies; climate change and marine pollution, e.g., ocean plastic pollution and oil spills. Here, we explore the effects of harmful fisheries subsidies in three marine ecosystems chosen for their importance in terms of food security, size and diversity (Mauritanian EEZ, South China Sea and East China Sea); and three different management scenarios, i.e., optimizing (i) economic rent; (ii) jobs; and (ii) ecological fitness. This rich modelling exercise allows us to address pertinent questions such as how much resource depletion is due to subsidies; how much rent depletion occurs due to the provision of harmful subsidies; and to what extent do differences in management regimes matter?

1. Introduction
At 70% of Earth's surface, the ocean is simply too big to mess up. But size is not the only reason we cannot afford to degrade the ocean, or even reduce it to a 'dead' ocean with little or no life, the ocean is actually a crucial natural asset that together with soils and forests, make up the world's stock of foundational natural capital (Lange et al. 2021; World Bank 2017). It is estimated that around 100 million tonnes of fish and invertebrates are caught annually from the global ocean (Pauly and Zeller, 2016), which, assuming an average ex-vessel price per tonne of $1,500 (Tai et al. 2017) and average economic impact multiplier of three (Dyck and Sumaila, 2010), generates total annual global gross revenues and economic impacts of $150 billion and $450 billion per year, respectively. Marine fisheries are therefore vital for the livelihoods of tens of millions worldwide, contributing directly to the food and nutritional security of billions of people (Golden et al. 2021), especially in the least developed coastal countries of the world, where, according to the United Nations Food and Agricultural Organization (FAO), supplies up to 20% of the animal protein that people consume. These fisheries also contribute indirectly to the well-being of people by serving as a source of vitally needed jobs and incomes (Teh and Sumaila 2013).

One of the essential ecosystem services that the ocean offers is climate regulation. Particularly, the ocean is absorbing 90% of the additional heat energy (von Schuckmann et al., 2020) and between 20-30% of the carbon dioxide (Friedlingstein et al., 2021) generated from anthropogenic sources of greenhouse gas emissions. Coastal blue carbon ecosystems such as mangroves, seagrasses remove and store carbon. Below-ground carbon storage in vegetated marine habitats is estimated to be up to 1000 tC ha–1, much higher than most terrestrial ecosystems (Bindoff et al. 2019). In addition, marine organisms in the upper ocean, from phytoplanktons to fish and marine mammals, contribute to sequestration and storage of carbon to the deep ocean through the biological pump. However, over-exploitation, habitat disturbance and losses from human activities release the stored carbon in ocean and coastal ecosystems and reduce their capacity to sequester and store carbon (Gattuso et al. 2018).

For this study, we selected three ecosystems based on their importance and the availability of previously constructed Ecopath with Ecosim models (Pauly et al. 2000) that can be updated and modified for our purposes (Cheung 2007; Guenette 2014). The fisheries based on these ecosystems are very important for Mauritania, and the countries bordering the ECS and SCS.

The fisheries active in Mauritania's EEZ caught an estimated 1.5 million tonnes of fish in 2018 and this generated an estimated $1.5 billion and $2.25 billion gross revenues and economic impact, respectively. The fisheries active in ECS caught approximately 5 million tonnes of fish in 2018 and this generated an estimated $8 billion and $23 billion gross revenues and economic impact, respectively. In the case of fisheries active in SCS, they caught about 11 million tonnes of fish in 2018 and this generated an estimated $16 billion and $47 billion gross revenues and economic impact, respectively.

The numbers in the preceding paragraphs underlie the importance of ensuring that we have fisheries that are economically viable through time (Sumaila 2021). However, many aspects of current ocean resource use patterns make it unsustainable due mainly to human transformation of marine ecosystems resulting in widespread biodiversity loss and habitat damage (Brondizio et al. 2019). Overfishing, destructive fishing practices, direct habitat damage, climate change and pollution are major anthropogenic threats to the future sustainability of oceans and their resources (Diaz et al. 2019; Lau et al. 2020). The consequences of these multiple stressors are the finding by the FAO that currently 34% of the world's fish stocks are overfished. However, others have estimated a larger proportion of depleted stocks, e.g., the Global Fish Index estimates that about 50% of marine fish stocks are overexploited.

Several reasons have been advanced in the literature for the current dismal state of global fisheries, including, ineffective management, harmful fisheries subsidies, illegal, unreported and unregulated (IUU) fishing; pollution (plastic, marine debris), below scientifically recommended MPA size (Sala et al. 2018), the common property nature of fish stocks, etc. (Gordon 1954; Pauly et al. 2002; World Bank 2017). Here, our focus is on harmful subsidies that reduce the private cost of fishing and thereby artificially inflate profits. Such subsidies generate excess fishing activity which then results in both biological and economic waste, impacting the sustainability of global fish stocks and reducing the net benefits generated from the fisheries for society. The excess capacity is also a major source of IUU fishing activities (Agnew et al. 2009; Sumaila et al. 2020).

Despite this general recognition, global action against harmful fisheries subsidies has been sluggish, as reflected in the slow progress of the WTO negotiations on fisheries subsidies, which started in 2001, and it is yet to reach an agreement. The next WTO Ministerial Meeting is scheduled for the week of June 12, 2022, and disciplining harmful subsidies is a top priority for the meeting.

Here, we conduct an analysis using ecosystem models and optimizations routines to investigate and seek answers to the following six policy-relevant and topical questions. Our goal is to produce additional information about the economic, social and ecological effects of harmful fisheries subsidies in support of science-based policymaking:

1. How much resource depletion is due to subsidies?
2. How much rent depletion occurs due to the subsidy?

3. To what extent do differences in management regimes matter?
4. How much of these differences in results are driven by differences in the magnitude of the subsidy?
5. To what extent does the ecology of the fishery matter in determining what is a less harmful subsidy?
6. What are the distributional trade-offs from removing subsidies?

2. Background information on the three ecosystems studied

2.1. Mauritanian EEZ

The Mauritanian EEZ is about 33,224 km^2 with a depth of less than 200m. There is a marine protected area in the EEZ, the Banc d'Arguin National Park, covering about 6,450 km^2 (Guenette et al. 2014). The shelf ecosystem of the Mauritanian EEZ is enriched by upwelling with diverse fish resources. The domestic fishery of Mauritania consists mainly of small-scale, artisanal fishing enterprises (Gascuel et al. 2007) that fish for mullets, especially in the Banc d'Arguin National Park where only Park residents with small sailboats are allowed to fish (Guenette et al. 2014). A large proportion of catches in the Mauritanian EEZ are caught by foreign vessels consisting mainly of pelagic and demersal fleets originating from the European Union, Russia, China and other countries (Gascuel et al. 2007).

2.2. East China Sea

The East China Sea (ECS) is a typical epicontinental large marine ecosystem (LME), covering about 770,000 km^2, bordering China, Japan and Korea (Li et al. 2009). The ECS is rich in nutrients with the influences of the alongshore current, the Yellow Sea cold water mass and the Kuroshio Current, providing the diverse fauna and flora in the ECS. In the past decades, the fishery resources in the ECS degraded because of overexploitation. The main fishing countries in the ECS are China, Japan and Korea. The landings in the ECS increased from less than 10 million tonnes in the 1950s to 45 million tonnes in the 2000s, with increasing fishing effort (Li et al., 2009). The ECS ecosystem experienced "fishing down the food web", i.e., the tendency to catch the biggest most valuable fish first then the next most valuable next, etc. (Pauly et al. 1998), and reached peak catch in 2013 (Sumaila 2019). For e.g., the mean trophic level of landings in the ECS decreased from 3.5 to 2.8 between 1965 and 1990 (Chao et al., 2005). The proportion of the total landings that is demersal fish[1] is decreasing over time while those for juveniles, small size and lower trophic level species are increasing – a clear indication of overall decline in the health of the fish stocks of the East China Sea.

2.3. South China Sea

The northern South China Sea (NSCS) is a typical coastal ecosystem, including all of the most diverse habitats that make up the South China Sea. There are many important spawning and fishing grounds in the NSCS that are rich with diverse fish resources. The fish resources in this part of the SCS have declined significantly due to the rapid increase in fish efforts over the last decades (Jia et al., 2005; Cheung, 2007; Sumaila and Cheung 2015; Sumaila et al. 2021). The fish community in the NSCS has therefore changed, with the fish species that were traditionally fished (e.g., the groupers) depleted and replaced by smaller lower tropic fish species (Cheung and Sadovy, 2004; Jia et al., 2005).

---

[1] That is fish living close to the sea floor, contrasted with pelagic fish that live closer to the surface.

3. Methods

3.1. Ecosim policy search scenarios

To address the objectives of this paper, we develop and apply Ecopath with Ecosim Models (ecopath.org) of the EEZ of Mauritania and the ecosystems of the ECS and SCS. These models are then run under the three different management policy scenarios specified:

- Economic optimization, where economic rents are maximized;
- Ecological optimization, where the biomass of the ecosystem, with particular focus on longer-lived species, is maximized; and
- Social optimization, where the number of jobs in the sector are maximized. This can be considered as a modelling metaphor for an open access regime where the aim is to maximize (labour) effort and by extension the catch, in the short term and neglecting other considerations.

We run the above scenarios 'with' and 'without' subsidies in order to isolate the effects of harmful subsidies on the following key indicators of a fishery: (i) fishing effort[2]; (ii) biomass; (iii) economic rent; and (iv) catch.

3.2. Ecopath with Ecosim models

A brief description of the EwE modelling framework is given in Supplementary Material (Supp. Mat. 1). Also, more detailed description of the theoretical basis of EwE and its applications can be found in Pauly et al. (2000); Sumaila (2004), Cheung et al. (2007); Cheung and Sumaila (2008); Sumaila et al. (2021).

3.2.1. Mauritanian EEZ

The Mauritanian EwE model applied in this study is adapted from the model of Guenette et al. (2014). The Mauritanian Ecopath model is based on the year 1991 and contains 51 functional groups (including 25 fish functional groups, 1 marine mammal, 1 seabird, and invertebrates). The fish functional groups were classified by habitat preferences, such as coastal, shelf, pelagic, and migratory. The fishing vessels in the model were divided into artisanal, demersal, and pelagic fleets. The catch of the pelagic fleets is about 10.77 tonnes/km^2, more than 80% of the total catch in Mauritania. The main targeted species of the pelagic fleets are large pelagic fish, Sardinellas, and horse mackerels. The basic parameters of the model are taken from the model of Guenette et al. (2014). The Ecosim model was fitted using functional group biomass time series data, spanning the period from 1991 to 2006.

3.2.2. East China Sea

The ECS model employed is an adapted EwE model of the 1970s built by Li and Zhang (2012). The model has 38 functional groups, including major fisheries resources and the important groups in the ecosystem, such as marine mammals and seabirds. The fishing vessels active in the ecosystem were divided into 12 fleet groups. The landings and bycatch data were obtained from the *Sea Around Us* (SAU) (Pauly & Zeller, 2016). The vulnerability index (*V*) values were taken from the estimates reported in Li and Zhang (2012).

---

[2] Fishing effort is the term used to measure the aggregate amount of inputs used for fishing and hence proxies the amount of fishing. A composite indicator may be used relating to a given combination of inputs, such as the number of hours or days spent fishing, numbers of hooks used (in long- line fishing), kilometres of nets used, etc.

However, we limited the V to a maximum of 10 to avoid unrealistic population growth in our projections and scenario analysis. The time series of fishing effort deployed in our ECS model were calculated based on the initial data of the 1970s ECS model and data presented in Cao et al. (2017) of the Chinese fleet vessel numbers, and average horsepower from 1995 to 2014. The growth rate of fishing efforts was estimated to be 0.65% per year (2000-2014).

3.2.3. South China Sea

The NSCS model we used in this project was built by Cheung (2007) based in the 2000s. The model covers the continental shelf (i.e., areas less than 200 m depth) of the NSCS, ranging from 106°53'-119°48' E to 17°10'-25°52' N. The coverage of the model mainly falls in the EEZ of China. The species were aggregated into functional groups based on the commercial importance, body size, ecology and the available data of the species (Cheung, 2007). There are 38 functional groups in the NSCS model, including 2 primary producers, 10 invertebrates, 21 fishes, 2 marine mammals, 1 marine turtle and 1 seabird group. The fishing vessels were defined into six fleets: pair and stern trawls, shrimp trawl, purse seine, hook and line, gillnet and others. The parameters of the NSCS model were evaluated based on government surveys, published literature, empirical equations and global databases. The vulnerability parameters of the model were transferred from the 1970s NSCS model, which was fitted by using the standardized CPUE data from 1973 to 1988 for 17 commercially exploited taxa to determine the vulnerability parameters (Cheung, 2007). The NSCS model can be used to predict the policy change impact on the ecosystem, particularly the fish species/functional groups in the NSCS.

3.3 Economic data, models and analysis

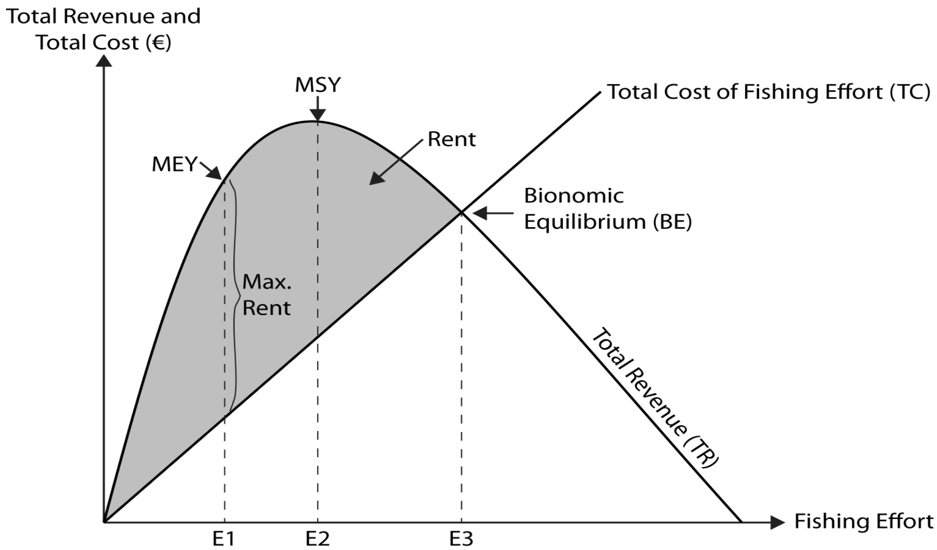
A brief description of the economic variables and parameters employed in this study are given in Supp. Mat. 2. Most of the information employed were obtained from the works of the FERU and the SAU, including databases on fishing costs (Lam et al. 2011), ex vessel prices (Tai et al. 2017), catch data (Pauly, Zeller, and Palomares 2020) and subsidies (Sumaila et al. 2019; Schuhbauer et al. 2020).

We compute the variable fishing cost and profit as the percentage of total revenues (or landed values) for each fishing fleet type in each of the models. These percentages are then used as input parameters for the economic conditions of the model. Fishing cost, ex-vessel price, landed value Sumaila et al. 2007; Swartz, Sumaila, and Watson 2012; Tai et al. 2017 and fisheries subsidies data are based on various databases developed by SAU (Pauly, Zeller, and Palomares 2020) and the Fisheries Economic Research Unit (Lam et al. 2011; Tai et al. 2017; Schuhbauer et al. 2019; Sumaila et al. 2019; Lange et al. 2021) at the University of British Columbia. We extracted the variables for different species or gear types in the model for each of the ecosystems being studied. Further details about economic models and data deployed in the current analysis are given in Supp. Mat. 2.

3.3.1 Fisheries subsidies

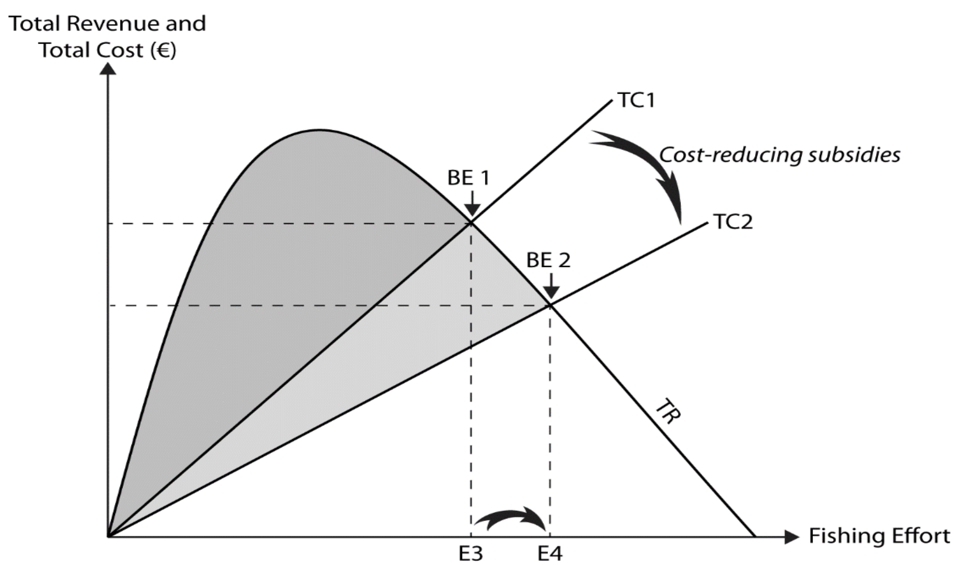
There are several variations of the definition of fisheries subsidies in the literature often provided by entities such the WTO (2021), Organisation for Economic Cooperation and Development (OECD) (Cox, 2006), the Food and Agriculture Organization of the United Nations (FAO, 1995; Westlund, 2004), UNEP (Abaza, 2002; Porter, 2004), Asia-Pacific Economic Cooperation (APEC, 2000), the World Bank (Milazzo, 1998), and the wider academic community (Sumaila et al. 2010; Sakai et al., 2019]) The common thread throughout each definition, as applied to the fisheries sector, is that a subsidy is a direct or indirect financial transfer from public entities that creates a benefit for the fisheries sector, which enable enterprises to make more profit than they would have otherwise (Sakai et al., 2019).

According to Gordon (1954), in open-access fisheries or common pool fisheries that are effectively managed, effort will continue to increase even when revenues per unit of effort are declining, and that ultimately revenues will decline until they equal costs. The point at which total revenue equals total cost is commonly regarded as the bionomic equilibrium (BE), where both industry profits and resource rents are completely dissipated (Figure 1). With subsidies, the fishing effort can actually exceed E3 (Figure 2; Sumaila 2002).



Source: Adapted from Sumaila and Pauly (2006).

Figure 1: Gordon-Schaefer bioeconomic model (Gordon 1954). This model describes the different parameters commonly used in bioeconomics theory.



Source: Adapted from Sumaila and Pauly (2006).

Figure 2: Effect of cost-reducing subsidies on fishing effort. This figure demonstrates that subsidies lowering the cost from TC1 to TC2, will also lower the bionomic equilibrium from BE1 to BE2, thus encouraging the growth of fishing effort from E3 to E4, hence the name 'capacity-enhancing' subsidies.

The latest global estimate of fisheries subsidies determines that USD 35.4 billion were provided by maritime countries in 2018, of which capacity-enhancing or harmful subsidies are USD 22.2 billion (Sumaila et al. 2019). The top five subsidising political entities (China, European Union, USA, Republic of Korea and Japan) contribute 58% (USD 20.5 billion) of the total estimated subsidy. For the three ecosystems studied in the current study, i.e., the Mauritanian EEZ, South China Sea and East China Sea, the number of harmful subsidies given to the vessels fishing in these ecosystems are USD 264 million, USD 990 million and USD 2,500 million in 2018, respectively. Note that harmful subsidies include those for fuel; fishing access agreements; boat construction and renewal; fisheries development projects; fishing port development; tax exemptions and marketing and storage infrastructure.

Two major issues harm the productivity and sustainability of global fisheries. The first is the open-access nature of most fisheries (or more generally their ineffective management), which leads to unsustainable levels of effort and fishing. That resource rents that are untaxed can be considered an "implicit subsidy" to fishers, similar to an untaxed externality, since social costs are generated alongside the private benefits generated. This type of subsidies was studied at the global level in the *Sunken Billions* Report and in academic papers. The second issue are large explicit subsidies in the fisheries sector, which increase fishing effort even above the already unsustainable open-access or effective management levels. This methods section focuses on these explicit subsidies and examines the distortions that they create both in isolation as well as in combination with different management scenarios.

4. The results

The results reported are for the net changes 'without' and 'with' subsidies. The first set of results presented in Tables 1-4 are based on 2018 estimated subsidy amounts that are currently provided to the fishing fleets active in each of the fisheries under study (Sumaila et al. 2019). This is followed by results that are computed by assuming that the same magnitude of subsidies in terms of percent of landed value are provided across the three ecosystems.

We present results for four key economic indicators in the main text of the report for the aggregate ecosystems. While providing aggregate numbers allows us to have a more focused presentation, we lose important details. Hence, we have provided fishing sector by sector results in Supp. Mat. 3.

4.1 How much fishing effort is stimulated due to subsidies?

In the next sections, we present the results from our modelling exercise for the fisheries of the Mauritanian EEZ, and the large marine ecosystems of the ECS and SCS. These are presented for three management scenarios: economic rent optimization, which mimics the maximum economic yield outcome; the job optimization scenario, which is close to the open access outcome; and the scenario that seeks to optimize ecology. In reality, the latter scenario mimics the implementation of mandated conservation policies such as the Endangered Species and the Species at Risk Acts of the United States and Canada, respectively.

Overall, the simulated fishing effort levels by 2030 across most fishing fleets in the Mauritania, South China Sea and East China Sea were estimated to be substantially higher than the baseline (2018) level when the systems were optimized for ecological benefits under both of the scenarios with and without subsidies. Simulated changes in fishing effort levels optimized for economic and social benefits were more variable when optimized for social benefits. In Mauritania, the "Artisanal" and "Pelagic" fleets were found to reduce fishing effort by 35 to 95% by 2030 relative to 2018 when the system was optimized for economic or ecological benefits. The simulated changes in fishing effort of demersal fleets were more variable (+/- 4 and 87% when optimized for economic and ecological benefits, respectively). In the South China Sea, fishing effort was shown to decrease by 50 to 80% by 2030 relative to 2018 when optimized for ecological benefits. In contrast, simulated changes in fishing effort range from +180% to -32% across different fishing fleets when optimized for economic and social benefits (Supp. Mat. 3). The simulated changes in fishing effort for the East China Sea varied widely between fishing gears. In general, demersal fishing gears such as bottom trawling increased while pelagic and mid-water fishing gears such as pelagic trawl, purse seine and gillnets decreased under economic and ecological optimization. Fishing effort generally increased when social benefits were optimized.

Comparing between the scenarios without subsidies relative to with subsidies, fishing efforts were generally lower without subsidies when the three ecosystems were optimized for economic and social benefits. However, differences in fishing effort levels optimized for ecological benefits between the two scenarios were more variable. When harmful subsidies are removed under the economic optimization scenario, fishing effort decreases across all the fleets by an average of 31% relative to when harmful subsidies are provided in the case of Mauritania fisheries while it decreases by 23% and 14% for the South China Sea and East China Sea, respectively. When optimizing for jobs, we see more variable changes in fishing effort. Fishing effort decreases for all fleets for Mauritania, but varies from a decrease of 42 and 30% to an increase of 35 and 9% for SCS and ECS, respectively. Optimizations for ecological benefits simulated the most variable differences in fishing efforts between the scenarios with and without harmful subsidies.

The optimizations of the models for the three case studies indicate some degree of “ecological engineering” i.e., manipulation of the ecosystems to design, construct or restore, and manage ecosystems that integrate human society with its natural environment for the benefit of both. Under economic optimization, the simulated changes in fishing effort are largely driven by economic principles. Therefore, fishing fleets with higher catch or value per cost are favoured, and that increases in fishing cost from removal of subsidies reduce fishing effort. In contrast, under optimization for jobs and ecological benefits, the models attempt to “engineer” the ecosystem and fisheries to maximize these benefits. For example, in the cases of the Mauritania and East China Sea, the optimized fishing indicated decrease in effort level for pelagic fleets to restore the biomass of long-lived pelagic species for ecological benefits. Simultaneously, demersal fishing gears that included valuable fast-growing invertebrates increased in effort to generate revenues. Similar simulation results, but under different ecosystems and fleet structures, can also explain the large variability in changes in effort across fleets for the two other ecosystems under job and ecological benefit optimizations. Such results would likely change if other ecological objective functions that account for intrinsic/non-market value of specific ecological groups are included in the optimization (Beattie et al. 2002).

Table 1: Effects of harmful subsidies on fishing effort management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies. The values represent minimum and maximum differences in fishing effort across fleets between the scenarios and the mean values (in parenthesis).

Indicator	**Optimizing economic rent (Net change without harmful subsidies)**	**Optimizing jobs (Net change without harmful subsidies)**	**Optimizing ecology (Net change without harmful subsidies)**
Mauritania	-7 to -75% (-31%) %	-6 to -30% (-9%)	-2 to +46% (+3%)%
SCS	-22 to -25% (-23%)	-42 to +35% (-5%)	-5 to 28% (+17%)
ECS	-40 to 28% (-14%)	-30 to 9% (-17%)	-11 to 20% (-3%)

4.2 How much resource depletion is due to subsidies?

To reveal how subsidies are likely to affect the biomass in an ecosystem (in which various species interactions such as predator-prey relations are present), we report in Table 2 the biomass of invertebrates and fishes separately. For instance, in the case of the SCS, the biomass of invertebrates decreases by 1% while those of fish species increase by 8% (Table 2). This decrease in the biomass of invertebrates is likely due to increases in predation pressure because of the increased abundance of fish species (see section 4.1. above). A key observation from our results is that when economic rent and jobs are optimized, subsidies reduce fish biomass overall but, in some cases, they do not impact invertebrates and fishes in the same way (see section 4.1. above). On the other hand, when the objective is to optimize the ecology, subsidies do not seem to have a marked impact on the biomass. This latter result is not surprising since protecting ecology is the goal.

Table 2: Effects of harmful subsidies on biomass by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	**Optimizing economic rent (Net change without harmful subsidies)**	**Optimizing jobs (Net change without harmful subsidies)**	**Optimizing ecology (Net change without harmful subsidies)**
Mauritania	+2%; +1%*	+6%; +7%	~0%; ~0%
SCS	-5%; +27%	-1%; +8%	~0%; -1.2%
ECS	+4%; -2%	+4%; +13%	~0%; ~0%

*First number is for invertebrate groups; the second number is for fish groups.

4.3 How much rent depletion occurs due to the subsidy?

Looking at the results reported in Table 3, it is clear that the removal of harmful subsidies results in large increases in rent, and vice versa. When optimizing economic rent, the removal of harmful subsidies results in a 9% (ESC) to 213% (SCS) increase in economic rent depending on the ecosystem (see section 4.1. above). The equivalent numbers when jobs and ecology are optimized are -3% (SCS) to 52% (Mauritania) and -5% (ECS) to 27% (SCS), respectively (Table 3).

Table 3: Effects of harmful subsidies on economic rent by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	Optimizing economic rent (Net change without harmful subsidies)	Optimizing jobs (Net change without harmful subsidies)	Optimizing ecology (Net change without harmful subsidies)
Mauritania	+10%	+52%	+7%
SCS	+213%	-3%	+27%
ECS	+9%	+49%	-5%

4.4 *How much catch is lost due to subsidy?*

From the point of view of the catch, we see that, even though fishing effort declines (Table 1), when economic rent is maximised, the quantity of catch taken either does not change (SCS and ECS) or decreases by 8% in the case of Mauritania. The decrease in biomass when jobs are maximized range between 2% and 7% (Table 4). When ecology is optimized, the relative catch changes are minor for Mauritania and the ECS. However, the biomass increases by 11% in the case of the SCS.

Table 4: Effects of harmful subsidies on catch by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	Optimizing economic rent (Net change without harmful subsidies)	Optimizing jobs (Net change without harmful subsidies)	Optimizing ecology (Net change without harmful subsidies)
Mauritania	-8%	-6%	-1%
SCS	~0%	-7%	+11%
ECS	~0%	-2%	-2%

4.5 *How much of the differences in the effect of harmful subsidies are driven by differences in the magnitude of the subsidy?*

To address the question posed in this section, we assumed that harmful subsidies are the equivalent of 30% of the landed value generated by each fishery and that these are removed. We also run simple sensitivity analysis using 10% and 50% of the landed value generated by each fishery to see how sensitive the results are relative to the magnitude of subsidies. The absolute amounts of subsidies equivalent to 10%, 30% and 50% of landed value by ecosystem by management scenario are given in Table 5.

Table 5: Subsidy amounts used in sensitivity analysis

Mauritania					
Sector	**2018 subsidy to LV ratio**	**2018 subsidy amount (USD M.)**	**Subsidy at 10% LV (USD M.)**	**Subsidy at 30% LV (USD M.)**	**Subsidy at 50% LV (USD M.)**
Artisanal	0.093	5.03	5.41	16.23	27.04
Demersl	0.098	15.16	15.47	46.41	77.35
Pelagic	0.122	114.91	94.19	282.57	470.94
Total		**135.1**	**115.07**	**345.20**	**575.33**

South China Sea					
Sector	**2018 subsidy to LV ratio**	**2018 subsidy amount (USD M.)**	**Subsidy at 10% LV (USD M.)**	**Subsidy at 30% LV (USD M.)**	**Subsidy at 50% LV (USD M.)**
Shrimp Trawl	0.185	246.57	133.28	399.84	666.41
Purse seine	0.19	102.24	53.81	161.43	269.05
Hook and line	0.187	25.75	13.77	41.31	68.85
Gillnet	0.19	117.89	62.05	186.14	310.24
Others	0.187	173.42	92.74	278.21	463.69
Total		**665.87**	**355.65**	**1,066.94**	**1,778.23**

East China Sea					
Sector	**2018 subsidy to LV ratio**	**2018 subsidy amount (USD M.)**	**Subsidy at 10% LV (USD M.)**	**Subsidy at 30% LV (USD M.)**	**Subsidy at 50% LV (USD M.)**
Gillnet	0.149	100	66.93	200.78	334.63
Hand tools	0.169	182	108.00	324.01	540.01
Line	0.176	75	42.50	127.51	212.52
Other industrial gears	0.173	72	41.67	125.01	208.35
Other nets	0.182	42	23.07	69.21	115.35
Other SS gears	0.032	48	148.64	445.93	743.22
Pelagic trawl	0.180	79	43.77	131.31	218.85
Purse seine	0.182	499	274.62	823.85	1,373.08
SS line	0.032	57	176.51	529.54	882.57
SS net	0.032	253	783.47	2,350.42	3,917.37
SS Pelatic trawl	0.032	46	142.45	427.35	712.25
Total		**1,453**	**1,852**	**5,555**	**9,258**

Table 6: Effects of removing the equivalent of 30% of the landed value of harmful subsidies on fishing effort by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	**Optimizing economic rent (Net change without harmful subsidies)**	**Optimizing jobs (Net change without harmful subsidies)**	**Optimizing ecology (Net change without harmful subsidies)**
Mauritania	-148%	-13%	+3%
SCS	-60%	-11%	+8%,
ECS	-55%,	-40%	-185%,

We see from Table 6 that under both economic and job optimization management objectives, fishing effort decreases by a significant amount in all fisheries, with larger decreases in economic optimization than when jobs are maximized. However, when ecology is optimised, only the fishing effort in the ECS decreases when harmful subsidies are removed while effort increases by 3% and 8% for the Mauritanian EEZ and SCS, respectively. See Section 4.1. for explanations that may explain why these results are obtained.

From Table 7, we observe that removing harmful subsidies leads to increases in biomass overall except when ecology is optimized, in which case the changes in biomass in the EEZ of Mauritania and the SCS hardly change while those of the ECS increase by over 20% (Table 7). Removing harmful subsidies leads to an increase in biomass of both invertebrates and fish species in the Mauritanian EEZ when optimizing for both economic and social management objectives. The results generated by the SCS model is different to the Mauritanian model in that the biomass of fish species decrease while those of invertebrates increase but by a much smaller percentage under the economic and social management objectives (see section 4.1. above). In the case of the SCS, we see a big percentage increase in the biomass of invertebrates while fish biomass barely changes when economic rent is optimised. However, under job and ecological optimization management objectives, the biomass of both invertebrates and fish species increases with the removal of harmful subsidies.

Table 7: Effects of removing the equivalent of 30% of the landed value of harmful subsidies on biomass by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	Optimizing economic rent (Net change without harmful subsidies)	Optimizing jobs (Net change without harmful subsidies)	Optimizing ecology (Net change without harmful subsidies)
Mauritania	+12%; +11%*	+6%; +6%	-0.1%; +0.1%
SCS	+34%; -7%	+9%; -0.4%	-0.1%; -1%
ECS	-0.4%; +71%	+17%; +7%	+22%; +26%

*First number is for Invertebrates; the second number is for Fish.

Table 8 presents results that show that economic rent increases by between 3% to 97% under economic rent and job optimization, depending on the ecosystem/fishery. When ecology is optimized, we see an increase in profit in the Mauritania and ECS fisheries and a decrease in the case of the SCS fisheries. The ecological optimization is driven largely by the biomass responses instead of the economics of fishing. Therefore, changes in economic rent under ecological optimization has very weak connection to the cost of fishing and thus the effects of scenarios with and without subsidies. See Section 4.1. for possible explanations for these results.

Table 8: Effects of removing the equivalent of 30% of the landed value of harmful subsidies on economic rent by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	Optimizing economic rent (Net change without harmful subsidies)	Optimizing jobs (Net change without harmful subsidies)	Optimizing ecology (Net change without harmful subsidies)
Mauritania	+49%	+19%,	+17%
SCS	+97%	-40%	-20%
ECS	+3%,	+25%	+84%

When it comes to the impact of removing subsidies of the equivalent of 30% of landed value on catch, we see in Table 9 that catches decrease by a much lower percentage compared to the decrease in fishing effort (Table 6) and economic rent (Table 8).

Table 9: Effects of removing the equivalent of 30% of the landed value of harmful subsidies on catch by management scenario by fishery expressed as net percentage change 'without' relative to 'with' subsidies

Indicator	Optimizing economic rent (Net change without harmful subsidies)	Optimizing jobs (Net change without harmful subsidies)	Optimizing ecology (Net change without harmful subsidies)
Mauritania	-6%	-6%	-2%
SCS	-0.5%	-7%	+0.4%
ECS	-12%	-12%,	-102%

4.6 *To what extent does the ecology of the fishery matter in determining what is a less harmful subsidy*

Given the results reported in Tables 5 - 8, we observe that when the equivalent of a 30% of landed value of subsidies is removed uniformly to all the fishing sectors, we can conclude that the ecology of the ecosystem upon which the fisheries are based matters. The type of species the ecosystem contains; the food web interactions between these species all matter. For example, Fisher and Mirman 1996) identified three types of interaction that would have implications for the kind of outcome one may encounter when fishing effort change for any reason, including due to the provision of harmful subsidies: (i) when the interaction is symbiotic, where having more of one species is good for another species; (ii) when species compete for a resource or are mutual predators; and (iii) when there is a predator-prey relationship between the species. In addition, the initial condition and state of the ecosystem and the fish stocks it contains at the beginning of the analysis matters. Whether a fishery is already effectively managed or not would also matter. We know that all the three ecosystems studied in our model are overfished (see Section 4.1).

4.7 *What are the distributional trade-offs from removing subsidies*

We address the question posed here using the Mauritanian model because the fishing fleets employed in the fishery are organized into three categories, i.e., the artisanal, demersal and pelagic fleets. This allows us to isolate the effects of removing harmful subsidies to all 3 fleets in one scenario, and doing so only to the demersal and pelagic fleets, i.e., only subsidies to the artisanal fleet are maintained. In particular, we are able to see what happens to the artisanal fleet where relatively small scale and poorer fishers operate.

Table 10a: No harmful subsidies to any of the three sectors. The results are expressed as net percentage change 'without' relative to 'with' subsidies.

	Fishing sector		
Management objective	**Artisanal**	**Demersal**	**Pelagic**
Optimizing economic rent			
Relative Rent without subsidies	-62%	28%	6%
Relative Fishing effort without subsidies	-75%	-7%	-6%
Optimizing jobs			
Relative Rent without subsidies	14%	3%	25%
Relative Fishing effort without subsidies	-7%	-6%	-29%
Optimizing ecology			
Relative Rent without subsidies	57%	3%	8%
Relative Fishing effort without subsidies	46%	-6%	-2%

Table 10b: Subsidies to artisanal sector only. The results are expressed as net percentage change 'without' relative to 'with' subsidies.

	Fishing sector		
Management objective	**Artisanal**	**Demersal**	**Pelagic**
Optimizing economic rent			
Relative rent, subsidies only artisanal fleet	22%	8%	6%
Relative effort, subsidies only artisanal fleet	16%	-15%	-6%
Optimizing jobs			
Relative rent, subsidies only artisanal fleet	3%	-22%	7%
Relative effort, subsidies only artisanal fleet	~0%	-3%	~0%
Optimizing ecology			
Relative rent, subsidies only artisanal fleet	18%	-11%	7%
Relative effort, subsidies only artisanal fleet	17%	-19%	-1%

Our results suggest that under economic rent optimization, removing harmful subsidies across all fisheries sectors at the same amount of 30% of landed values would lead to reductions in fishing effort in all sectors of between 6% and 75%, depending on the sector (Table 10a). However, the economic rent to the artisanal sector is the only one that declines by over 60% relative to the scenario where subsidies are provided (Table 10a). In fact, the demersal and pelagic sectors increase their economic rent by 28% and 6%, respectively. This means that removing subsidies across all sectors would be undesirable distributionally, with the artisanal fleet losing out big time. Future work that studies several diverse models from across the global ocean would help us determine whether this result is generalizable.

We re-run the Mauritanian EEZ with subsidies given to only the artisanal sector and report the results generated in Table 10b. We see from this table that under economic rent maximization, the net economic rent to the artisanal fleet increases from -62%, when harmful subsidies to all fishing sectors are removed, to 22% when only the artisanal fleet are given subsidies. The good news here is both the demersal and pelagic fleets still see increases in the economic rent under this subsidy scenario. Given that most of the pelagic and demersal fleets in Mauritania's EEZ are foreign owned, our results suggest that a possible 2nd best solution is to ban foreign fleets that are subsidised from fishing in the EEZ of Mauritania.

5. Policy implications and conclusion remarks

We have built EwE models for three diverse ecosystems capturing the current state of the fish and fisheries dependent on them, the economics of the fisheries as well as the social state of the fisheries. We have incorporated different levels of harmful subsidies provided to the fishing fleets active in this ecosystem, and arrived at a set of rich results that are backed by the economic theory of fisheries subsidies. Allowing for the ecology and how effective the ecosystems are managed initially, our results confirm the theoretical expectations that removing harmful subsidies would reduce fishing effort and overfishing while improving the economics, especially when the management objective is to optimize economic rent.

Overall, our results suggest that removing harmful subsidies would lead to increases in aggregate economic rent for the fisheries operating in all three ecosystems and for all three management objectives. The results also show that the biomass level would increase overall or at the minimum would not decline when harmful subsidies are removed. However, the path to achieving the increased profitability is diverse and rich in relation to fishing effort and aggregate catch.

In the case of Mauritania, our results suggest that the net economic gain is mainly achieved by cutting effort and reducing fishing cost under the economic rent maximization management regime. Costs are also reduced by boosting biomass, reducing catch and prioritizing the catch of fish that command higher prices in the market. Similar reasons can also be ascribed to the increase in economic rent when the management objective is to maximize jobs.

In the case of the gain in rent achieved in the South China Sea when economics is optimized, this seems to be due to a combination of decrease in fishing effort, increase in catch due to a rebound in fish stocks and an overall increase in biomass (see Section 4.1.). These helped to increase economic rent by both increasing revenues and reducing the cost of fishing. Rent increases when the social objective is maximized through small increases in catch, effort and biomass. When ecology is optimized, economic rents are enhanced via increases in both fishing effort and catch but this is done in a way that maintains the biomass (see explanations given in Section 4.1).

Overfishing as well as destructive fishing activities cause long-term damage to fish habitats and fish stocks, which result in shifts in ecosystem and fish community structures. There are many direct and indirect reasons for the unsustainability of our seas, oceans and fisheries. One major factor is overcapacity of world fishing fleets. The literature is replete with ideas on what good fishery policy looks like. To ensure sustainable fisheries, we need to deal with the problems of 'open access' and 'common property' that is not jointly managed. The provision of harmful fisheries subsidies and the elimination of IUU fishing are required. We need to re-establish the natural protection once afforded to fish by establishing marine protected areas. Most importantly, we need to break with short-sightedness, and value benefits in a manner that explicitly considers the interest of future generations.

Notwithstanding the fact that we've set up and run huge completed ecosystem models, our results are generally clear, eliminating harmful subsidies will lead to reductions in fishing effort in the economic and job maximization scenarios. The consequences of these reductions are to improve the profitability of the fisheries while improving fish biomass. One final important result is that removing subsidies under a job maximisation, which mimics open access) is still worthwhile as subsidy reform actually forces fishers to reduce fishing effort. In other words, removing subsidy without closing open access is not futile.

References

Abaza, H., E. Fellus (2002). Fisheries Subsidies and Marine Resource Management: Lessons Learned from Studies in Argentina and Senegal, UNEP.

Agnew, D. J., Pearce, J., Pramod, G., Peatman, T., Watson, R., Beddingtonne, J. R., & Pitcher, T. J. (2009). Estimating the worldwide extent of illegal fishing. *PloS one*, *4*(2), e4570.

APEC (2000). Study into the Nature and Extent of Subsidies in the Fisheries Sector of APEC Member Economies. https://www.apec.org/Publications/2000/10/Study-into-the-Nature-and-Extent-of-Subsidies-in-the-Fisheries-Sector-of-APEC-Member-Economies-2000 (accessed May 12, 2019).

Beattie, A., Sumaila, U. R., Christensen, V., & Pauly, D. (2002). A model for the bioeconomic evaluation of marine protected area size and placement in the North Sea. *Natural Resource Modeling*, *15*(4), 413-437.

Bindoff, N. L., Cheung, W. W., Kairo, J. G., Arístegui, J., Guinder, V. A., Hallberg, R., Hilmi, N. J. M., Jiao, N., Karim, M. S., & Levin, L. (2019). Changing ocean, marine ecosystems, and dependent communities. In IPCC special report on the ocean and cryosphere in a changing climate (pp. 477–587). Intergovernmental Panel on Climate Change.

Brondizio, E.S., Settele, J., Díaz, S., & Ngo, H.T. (eds.) (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. (IPBES secretariat, Bonn).

Cao, L., Chen, Y., Dong, S., Hanson, A., Huang, B., Leadbitter, D., & Little, D. C. (2017). Opportunity for marine fisheries reform in China. *PNAS*, *114*(3), 435–442.

Chao, M., Quan, W. M., Hou, L. C., & Chen, Y. H. (2005). Changes in trophic level of marine catches in the East China Sea region. *Marine Sciences*.

Cheung, W. W. (2007). *Vulnerability of marine fishes to fishing: from global overview to the Northern South China Sea*. (Doctor), University of British Columbia.

Cheung, W.W. and Sumaila, U.R., 2008. Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. *Ecological economics*, *66*(1), pp.193-210.

Cheung, W. W., & Sadovy, Y. (2004). Retrospective evaluation of data-limited fisheries: a case from Hong Kong. *Reviews in fish biology and fisheries, 14*(2), 181-206.

Christensen, V., & Walters, C. J. (2004). Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling, 172*(2-4), 109-139.

Cox, A. (2006). Financial Support to Fisheries: Implications for Sustainable Development, OECD Publishing.

Diaz, S., Settele, J., Brond'izio, E., Ngo, H., Guèze, M., Agard, J., Arneth, A., Balvanera, P., Brauman, K., Butchart, S., & Others. (2019). Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

Dyck, A. J., & Sumaila, U. R. (2010). Economic impact of ocean fish populations in the global fishery. *Journal of Bioeconomics*, *12*(3), 227-243.

Eez, M., Gascuel, Zeller, T., Sidi, & Pauly, D. (2007). *Reconstructed catches in the Mauritanian EEZ*: Reconstruction of marine fisheries catches for key countries and regions (1950-2005).

FAO, Code of Conduct for Responsible Fisheries.

G. Porter, Incorporating resource Impact into Fisheries Subsidies Disciplines: Issues and Options, Economics and Trade Branch Discussion Paper, UNEP, Geneva. (2004).

Friedlingstein, P., Jones, M. W., O'Sullivan, M., Andrew, R. M., Bakker, D. C. E., Hauck, J., Le Quéré, C., Peters, G. P., Peters, W., Pongratz, J., Sitch, S., Canadell, J. G., Ciais, P., Jackson, R. B., Alin, S. R., Anthoni, P., Bates, N. R., Becker, M., Bellouin, N., ... Zeng, J. (2021). Global Carbon Budget 2021. Earth System Science Data Discussions, 2021, 1–191. https://doi.org/10.5194/essd-2021-386

Gascuel, D., Labrosse, P., Meissa, B., Sidi, M. T., & Guenette, S. (2007). Decline of demersal resources in North-West Africa: an analysis of Mauritanian trawl-survey data over the past 25 years. *African Journal of Marine Science, 29*(3), 331-345.

Gattuso, J.-P., Magnan, A. K., Bopp, L., Cheung, W. W. L., Duarte, C. M., Hinkel, J., Mcleod, E., Micheli, F., Oschlies, A., Williamson, P., & Others. (2018). Ocean solutions to address climate change and its effects on marine ecosystems. Frontiers in Marine Science, 5, 337.

Golden, C. D., Koehn, J. Z., Shepon, A., Passarelli, S., Free, C. M., Viana, D. F., ... & Thilsted, S. H. (2021). Aquatic foods to nourish nations. *Nature*, *598*(7880), 315-320.

Gordon H.S., 1954. *The economic theory of common property resource: the fishery*. Journal of Political Economy 62, 124-143.

Guenette, S., Meissa, B., & Gascuel, D. (2014). Assessing the Contribution of Marine Protected Areas to the Trophic Functioning of Ecosystems: A Model for the Banc d' Arguin and the Mauritanian Shelf. *Annals of Geophysics, 46*(1), 1-26.

Jia, X., Li, C., & Qiu, Y. (2005). Survey and evaluation of Guangdong marine fishery resources and the measures for sustainable utilization. *Chinese marine publication*.

Lam, Vicky W Y et al. 2011. "Construction and First Applications of a Global Cost of Fishing Database." *ICES Journal of Marine Science: Journal du Conseil* 68(9): 1996–2004. http://icesjms.oxfordjournals.org/content/68/9/1996.abstract.

Lange, Glenn-Marie et al. (2021). Blue Natural Capital: Mangroves and Fisheries. In *The Changing Wealth of Nations 2021 - Managing Assets for the Future*, World Bank Group, 504.

Lange, G. M., Beck, M. W., Lam, V. W., Menéndez, P., & Sumaila, U. R. (2021). Blue Natural Capital: Mangroves and Fisheries.

Lau, W. W., Shiran, Y., Bailey, R. M., Cook, E., Stuchtey, M. R., Koskella, J., ... & Palardy, J. E. (2020). Evaluating scenarios toward zero plastic pollution. *Science*, *369*(6510), 1455-1461.

Li, Y., Yong, C., Olson, D., Yu, N., & Chen, L. (2009). Evaluating ecosystem structure and functioning of the East China Sea Shelf ecosystem, China. *Hydrobiologia, 636*(1), 331.

Li, Y., & Zhang, Y. (2012). Fisheries impact on the East China Sea Shelf ecosystem for 1969-2000. *Helgoland Marine Research, 66*(3), 371-383.

Milazzo, M. (1998). Subsidies in World Fisheries: A Re-examination, The World Bank.

Pauly, D., Christensen, V. V., Dalsgaard, J., Froese, R., & Jr, T. F. (1998). Fishing down marine food webs. *Science*, 279(5352), 860-863.

Pauly, D., & Zeller, D. (2016). Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining. *Nature communications*, *7*(1), 1-9.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J., ... & Zeller, D. (2002). Towards sustainability in world fisheries. *Nature*, *418*(6898), 689-695.

Pauly, D., Christensen, V., & Walters, C. (2000). Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. *ICES journal of Marine Science*, *57*(3), 697-706.

Pauly, D., D. Zeller, and M.L.D. Palomares. 2020. "Sea Around Us Concepts, Design and Data (Seaaroundus.Org)."

Porter, G. (2004). Incorporating resource Impact into Fisheries Subsidies Disciplines: Issues and Options, Economics and Trade Branch Discussion Paper, UNEP, Geneva.

Sakai, Y., N. Yagi, U.R. Sumaila (2019). Fishery Subsidies: The Interaction Between Science and Policy, Fisheries Science. 85 (2019) 439–447.

Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C., & Sumaila, U. R. (2018). Assessing real progress towards effective ocean protection. *Marine Policy*, *91*, 11-13.

Schuhbauer, Anna, Daniel J Skerritt, Naazia Ebrahim, and U Frédéric Le Manach (2019). The Global Fisheries Subsidies Divide between Small-and Large-Scale Fisheries.

Sumaila U.R., 2002. *Achieving sustainable fisheries: the economic dimension. Paper presented at the UNEP Workshop on Fisheries Subsidies and Sustainable Fisheries Management*, April 26-27, 2001, United Nations Environment Programme, Geneva (Switzerland).

Sumaila, U Rashid, Daniel Skerritt, et al. (2019). A Global Dataset on Subsidies to the Fisheries Sector. *Data in brief* 27: 104706.

Sumaila, U. R, Naazia Ebrahim, et al. (2019). Updated Estimates and Analysis of Global Fisheries Subsidies. *Marine Policy* 109: 103695.

Sumaila, U. R, A. D. Marsden, R. Watson, and D. Pauly (2007). A Global Ex-Vessel Fish Price Database: Construction and Applications. *J. Bioecon*. 9(1): 39–51.

Sumaila, U.R., A.S. Khan, A.J. Dyck, R. Watson, G. Munro, P. Tydemers, D. Pauly (2010). A bottom-up re-estimation of global fisheries subsidies, *Journal of Bioeconomics*. 12 (2010) 201–225.

Sumaila, U. R. (2021). *Infinity Fish: Economics and the Future of Fish and Fisheries*. Elsevier.

Sumaila, U. R. (2004). Intergenerational cost–benefit analysis and marine ecosystem restoration. *Fish and Fisheries*, *5*(4), 329-343.

Sumaila, U. R. (2019). Comparative valuation of fisheries in Asian large marine ecosystems with emphasis on the East China Sea and South China Sea LMEs. *Deep Sea Research Part II: Topical Studies in Oceanography*, *163*, 96-101.

Sumaila, U. R., & Cheung, W. W. (2015). Boom or bust: The future of fish in the South China Sea. *Vancouver, BC: University of British Columbia. [Available online at: https://drive. google. com/file/d/0B_oUJE4kCTZrbVI4N2tTVjlpYTA/view. Accessed 05 April 2018].*

Sumaila, U.R., Cheung, W. W. et al. (2021). Sink or Swim: The future of fisheries in the East and South China Seas. ADM Capital, Hong Kong.

Sumaila, U. R., Zeller, D., Hood, L., Palomares, M. L. D., Li, Y., & Pauly, D. (2020). Illicit trade in marine fish catch and its effects on ecosystems and people worldwide. *Science advances*, *6*(9), eaaz3801.

Swartz, Wilf, Rashid Sumaila, and Reg Watson. 2012. "Global Ex-Vessel Fish Price Database Revisited: A New Approach for Estimating 'Missing' Prices." *Environ. Resour. Econ.*: 1–14. http://dx.doi.org/10.1007/s10640-012-9611-1.

Tai, T.C., Cashion, T., Lam, V.W., Swartz, W. and Sumaila, U.R., (2017). Ex-vessel fish price database: Disaggregating prices for low-priced species from reduction fisheries. *Frontiers in Marine Science*, *4*, p.363.

Teh, L. C., & Sumaila, U. R. (2013). Contribution of marine fisheries to worldwide employment. *Fish and Fisheries*, *14*(1), 77-88.

United Nations (2002), Report of the World Summit on Sustainable Development, August 26 - September 4, 2002, Johannesburg (South Africa).

von Schuckmann, K., Holland, E., Haugan, P., & Thomson, P. (2020). Ocean science, data, and services for the UN 2030 Sustainable Development Goals. Marine Policy, 121(February), 104154. https://doi.org/10.1016/j.marpol.2020.104154.

Walters, C., Pauly, D., Christensen, V., & Kitchell, J. F. (2000). Representing density dependent consequences of life history strategies in aquatic ecosystems: EcoSim II. *Ecosystems, 3*(1), 70-83.

Westlund, L. (2004). Guide for Identifying, Assessing and Reporting on Subsidies in the Fisheries Sector, FAO.

World Bank. (2017). *The sunken billions revisited: Progress and challenges in global marine fisheries*. The World Bank.

WTO (2021). Negotiations on fisheries subsidies: https://www.wto.org/english/tratop_e/rulesneg_e/fish_e/fish_e.htm (Last accessed: 27/08/2021).

Appendices

Supp. Mat. 1: Ecopath with Ecosim model

The Ecopath with Ecosim model (EwE) is one of the more widely used ecosystem models, as a useful tool to evaluate the effects of fishing and fisheries management policies on ecosystems (Christensen and Walters, 2004). EwE can be used to describe the ecological process of an ecosystem. The Ecopath is a mass-balance model, representing the snap-shot of the ecosystem at a particular time period. Species in ecosystems are aggregated into functional groups to reduce the number of state variables, depending on their biological characteristics. The input parameters of the functional groups of the model, the biomass (B); production to biomass ratio (P/B); the consumption to biomass ratio (Q/B); and the ecotrophic efficiency (EE) can represent their biological characteristics (Christensen and Walters, 2004). There are two basic equations for the Ecopath model to govern the mass-balance principle of the ecosystem. The first equation (1) is the balance between production, consumption, predation, fishing, migration and other mortality among functional groups:

$$(P/B)_i \cdot B_i \cdot (1-EE)_i - B_j \cdot \left(\frac{Q}{B}\right)_j \cdot DC_{ji} - Y_i - E_i - BA_i = 0 \qquad (1)$$

The second equation (2) is the balance between consumption, production and respiration within a functional group:

$$Q_i = P_i + R_i + GE_i \cdot Q_i \qquad (2)$$

where $(P/B)_i$ is the production to biomass ratio of functional group i; B_i the total biomass of functional group i; EE_i is the ecotrophic efficiency of functional group i; Y_i is the total catch functional group i; E_i is the net migration functional group i; BA_i is the biomass accumulation of functional group i; $(Q/B)_j$ are consumption to biomass ratio for predator functional groups j; DC_{ji} is the proportion of the functional group i in the diet of predator functional groups j; R is respiration while GE is the proportion of unassimilated food (Christensen and Walters, 2004).

The Ecosim is a time-dynamic model, which provides scientists with a useful tool to explore the temporal dynamics of a food web under the effects of fishing, environmental changes and fishery management policies (Christensen and Walters, 2004). There are two equations of the Ecosim model for the simulation of biomass changes of ecosystem functional groups (Walters et al., 2000):

$$P_i = g_i \sum_j C_{ji} - \sum_j C_{ij} + I_i - (M_i + F_i + e_i)B_i \qquad (3)$$

and

$$C_{ij} = \frac{v_{ij} \cdot a_{ij} \cdot B_i \cdot B_j}{v_{ij} + v'_{ij} + a_{ij} \cdot B_j} \qquad (4)$$

where P_i is the net production of functional group i in terms of its biomass, g_i is the growth efficiency of functional group i, M_i and F_i are the natural and fishing mortalities of functional group i, I_i and e_i are immigration and emigration rates of functional group i, C_{ji} is the consumption of functional group $_j$ by group $_i$, v and v' parameters is rates of behavioural exchange between invulnerable and vulnerable states, and a_{ij} is the rate of effective search by predator $_j$ for prey type.

Supp Mat. 2: Economic data, models and analysis

We compute the variable fishing cost and profit as the percentage of total revenues (or landed values) for each fishing fleet type in each of the models. These percentages are then used as input parameters for the economic conditions of the model. Fishing cost, ex-vessel price, landed value and fisheries subsidies data are based on various databases developed by SAU (Pauly, Zeller, and Palomares 2020) and the Fisheries Economic Research Unit (Lam et al. 2011; Schuhbauer et al. 2019; Sumaila, Ebrahim, et al. 2019; Tai et al. 2017) at the University of British Columbia. We extracted the variables for different species or gear types in the model for each of the ecosystems being studied. Further details about economic models and data deployed in the current analysis are given in Supp. Mat.

2.3.1 Fishing cost

The unit variable fishing cost data (USD per tonne of catch) is extracted from the global fishing cost database (Lam et al. 2011; Lange et al. 2021). The variable fishing cost includes the fuel cost, labour cost, maintenance cost and other running cost when operating the fishing vessels. The cost data is aggregated by fishing country and fishing gear. The variable fishing cost for all fishing countries in each EEZ or ecosystem is extracted. By linking the fishing gear type to the fishing fleet, the fishing cost for each fishing fleet can be estimated. The variable fishing cost to the landed value ratio for each fishing fleet type is then calculated.

2.3.2 Ex-vessel price data and landed value

Ex-vessel prices are the prices that fishers receive directly for their catch, or the selling price when it first enters the supply chain. Ex-vessel price data (in USD per tonne of catch) of each taxon by year (from 2009 to 2018) and fishing country was extracted from the Fisheries Economic Research Unit (FERU) global ex-vessel price database (Sumaila et al. 2007; Swartz, Sumaila, and Watson 2012; Tai et al. 2017). The detailed method for building the global ex-vessel database can be found in Tai et al., 2017. The price is matched with each record of the catch data of each marine taxon caught by fishing country and gear type in the SAU reconstructed catch database. The gear types used in the SAU database are then matched with the fleet type in each model. For example, in Mauritania EEZ, the marine taxon groups caught by each fleet type (i.e., artisanal, pelagic and demersal) are matched with the marine species with price data in the SAU database. We then get the weighted average unit price of each fleet type used in each EEZ or ecosystem. By combining the catch data with the fishing ex-vessel price data of each exploited marine taxon, the landed values can be estimated for different fishing fleet types for each EEZ or ecosystem.

2.3.3 Fisheries subsidies

Two major issues harm the productivity and sustainability of global fisheries. The first is the open-access nature of most fisheries (or more generally their ineffective management), which leads to unsustainable levels of effort and fishing. That resource rents are untaxed can be considered an “implicit subsidy” to fishers, similar to an untaxed externality, since social costs are generated alongside the private benefits generated. This type of subsidies was studied at the global level in the *Sunken Billions* Report and in academic papers. The second issue are large explicit subsidies in the fisheries sector, which increase fishing effort even above the already unsustainable open-access or effective management levels. This methods section focuses on these explicit subsidies and examines the distortions that they create both in isolation as well as in relation to the open-access or ineffective management, implicit subsidies. We model the impact of these subsidies of the large fisheries, i.e., those active in the EEZ of Mauritania, and the large marine ecosystems of the South China Sea, and East China Sea.

Capacity enhancing subsidies data are extracted from the FERU subsidies database (Schuhbauer et al. 2019; Sumaila, Skerritt, et al. 2019). Capacity enhancing subsidies include boat construction and renovation, fisheries development projects, fishing ports development, market and storage infrastructure, tax exemption, fishing access and fuel subsidies. The subsidies allocated to the large-scale fishing (LSF) sector is larger than those allocated to the small-scale fishing (SSF) sector with 19% of global fishing subsidies going to small-scale fishing sector (Schuhbauer et al. 2019). However, the fleet types in each model do not distinguish between large-scale and small-scale fisheries. Therefore, we get the subsidy data for each functional group and then allocate the subsidies to each fleet type based on the functional group of the marine species that this fleet is targeting.

The capacity enhancing subsidies for each fishing sector is aggregated by the fishing country in the FERU subsidies database. We first estimate the capacity enhancing subsidies for LSF by each fishing country in the EEZ or ecosystem using the catch ratio (i.e., catch in this EEZ or ecosystem by each fishing country / total catch of this fishing country globally). Then, we estimate the subsidies for each functional group and fishing sector in each EEZ (i.e., Mauritania) or ecosystem (Northern South China Sea and East China Sea). We extract the catch of large-scale and small-scale fishing sectors from 2009 -2018 from the SAU database, respectively. The annual average catch for each functional group, fishing country and fishing sector is then calculated. We then calculate the LSF subsidies by each functional group using the catch proportion of each functional group by each country in the EEZ or ecosystem. We also extract the subsidies for SSF and calculate the SSF subsidies allocated to each functional group using the catch proportion.

We sum up the subsidy and landed values for all fishing countries harvesting in each EEZ or ecosystem. With the catch proportion of the functional group, we can estimate how much catch is allocated to SSF or LSF of each functional group for each fleet type. The functional group is then linked to the target species group of each fleet type. The catch amount of each target species group allocated to SSF or LSF is estimated based on the catch ratio of each group in each fishing sector from the SAU database. The total subsidy for each fleet type is therefore estimated by summing up the subsidy in the SSF and LSF sector. The final subsidy to landed values ratio of each fleet type is computed and this ratio is one of the input variables to the model.