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Diagnosis and Management of Small-Scale and Data-Limited Fisheries

Edited by
Mohamed Samy-Kamal and Célia M. Teixeira

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Guest Editors

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Cover image courtesy of Mohamed Samy-Kamal

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About the Editors

Mohamed Samy-Kamal

Dr. Samy-Kamal is a fisheries scientist specializing in fisheries management, regulations, and strategies, particularly in small-scale fisheries. He holds a Master's degree in Economics and Management of Fisheries from the University of Barcelona and a PhD in Marine Science and Applied Biology from the University of Alicante, Spain.

His research has focused on analyzing the effects of management measures on Mediterranean and Red Sea fisheries, as well as the legal frameworks and management systems of fisheries in countries such as Spain, Egypt, Russia, and Estonia. He has authored over 25 peer-reviewed scientific papers on these topics.

Dr. Samy-Kamal has taught and supervised master's theses at the University of Alicante and IAMZ-CIHEAM, and has served as a member of thesis dissertation committees. He has also reviewed over 70 papers for international journals.

Recently, Dr. Samy-Kamal has been involved in various Marine Stewardship Council (MSC) fishery assessments in diverse geographic areas, including the Adriatic Sea, Barents Sea, Lake Peipus, Chany Lake, Curonian Lagoon, Bratsk Reservoir, Parnu Bay, the Baltic Sea, and the North Pacific (Sea of Japan, Bering Sea, and Sea of Okhotsk).

Celia M. Teixeira

Dr Celia M. Teixeira's research is focused on the sustainability of fisheries, as well as the effects of climate change on fisheries and different issues regarding small-scale fisheries . She has authored or co-authored four books, one book chapter, and over 30 peer-reviewed scientific papers on these topics.

Since 2000, she has divided her career between scientific research and education. She was a lecturer at the Piaget Institute, and she was an invited lecturer on the topic "Fisheries Management" for the MsC Marine Ecology course at the Faculdade de Ciências da Universidade de Lisboa, and for the MsC in Management and Conservation of Natural Resources at the Universidade de Évora. Currently, she lectures in Marine Biology at ISPA. She has supervised MsC students, PhD students, and hired fellows. She has taken part in several juries for academic graduations. She has also reviewed over 70 papers for international journals.

Preface

Small-scale and data-limited fisheries, often overlooked in global fisheries discussions, are crucial to the livelihoods of millions worldwide. These fisheries, characterized by their limited scale, reliance on local knowledge, and a lack of comprehensive biological and ecological information, face numerous challenges, including overfishing, climate change, habitat degradation, and market fluctuations. Their unique characteristics also pose significant management challenges.

To ensure the sustainability of these valuable fisheries resources, effective diagnosis and management strategies are essential. This Special Issue aims to provide an overview of the key issues facing small-scale and data-limited fisheries and to offer practical solutions. By examining case studies from diverse regions, we hope to provide practical guidance for researchers, policymakers, and fisheries managers, and equip them with the tools and knowledge needed to address these complex issues. The chapters delve into a range of methodologies that can be applied to promote sustainable resource use and address the unique challenges posed by these fisheries.

We extend our sincere gratitude to all authors and reviewers who contributed to this Special Issue, generously sharing their expertise and insights. We also acknowledge the exceptional support and cooperation of the Managing Editors, Ms. Joyce Chen and Ms. Carola Wang, and the staff of Fishes. Their dedication and professionalism were invaluable in bringing this project to fruition.

This reprint is intended to serve as a valuable reference resource for students, fisheries researchers, analysts, policymakers, and managers. We hope it will fulfill its intended purpose of providing insights into the challenges and opportunities associated with small-scale and data-limited fisheries.

Mohamed Samy-Kamal and Célia M. Teixeira
Guest Editors

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Editorial

Diagnosis and Management of Small-Scale and Data-Limited Fisheries

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Historically, small-scale fisheries (SSFs) have largely been overlooked by fisheries scientists and management authorities at national and international levels. This disregard stems from a misperception and undervaluation of the socio-economic significance of SSFs' contributions to society's well-being. Although SSFs are sometimes disregarded or marginalized due to their poor economic value, they are essential for employment and may be economically valuable for locals. SSFs are estimated to be responsible for more than half of all landings globally, provide food security for millions of people worldwide, and employ more than 90% of all wild-catch fishers. SSFs are typically multi-gear and multi-species, play an important role in maintaining household and community livelihoods, and contribute considerably to the local and international trading of seafood products. This lack of attention has meant fewer resources assigned for data collection and the assessment of their stocks, especially for those stocks with relatively low commercial value. For this reason, most of the world's fish stocks are considered data limited. This also compromises the decision-making process and the implementation of adequate management measures and regulations when managers must make decisions in the absence of data and/or adequate scientific advice.

Many people associate the term SSFs with relatively small, traditional fishing boats equipped with low-technical gear and labor-intensive fishing methods. SSFs are especially difficult to define due to the usage of multiple criteria to describe them and the reusability of terminologies, such as small-scale fisheries, artisanal fisheries, subsistence fisheries, and traditional fisheries. Because of the considerable diversity of SSFs across the world, it is difficult to construct a widely agreed definition. The Food and Agricultural Organization (FAO) has defined small-scale or artisanal fisheries as "traditional fisheries involving fishing households (as opposed to commercial companies), using relatively small amount of capital and energy, relatively small fishing vessels (if any), making short fishing trips, close to shore, mainly for local consumption. In practice, the definition varies between countries, e.g., from gleaning or a one-man canoe in poor developing countries, to more than 20-m trawlers, seiners, or long-liners in developed ones. Artisanal fisheries can be subsistence or commercial fisheries, providing for local consumption or export".

In 2015, the FAO also created the "Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries" (hence referred to as the "SSF Guidelines") to raise awareness of SSFs and encourage states to fund projects for their sustainability [1]. In the SSF Guidelines, with an emphasis on disaggregated data that make SSF more visible to decision-makers, the FAO urges states to improve (or establish) data collection programs for SSFs. Section #11.1 of the guidelines makes the most explicit request for disaggregated data: "States [i.e., Nations] should establish systems of collecting fisheries data, including bioecological, social, cultural and economic data relevant for decision-making on sustainable management of small-scale fisheries with a view to ensuring sustainability of ecosystems, including fish stocks, in a transparent manner. Efforts should be made to also produce gender-disaggregated data in official statistics, as well as data allowing for an improved understanding and visibility of the importance of small-scale fisheries and its different components, including socioeconomic aspects".

The challenges and threats that face SSFs are rarely attributable to a single cause or problem. For example, international organizations, such as the FAO, routinely collect data on fisheries landings at the national level and aggregate that data at regional and global levels. However, in many cases (developing countries), there is frequently no distinction between landings from SSFs and larger-scale commercial ventures. The sustainable management of stocks is severely challenged by the insufficient catch, survey, and other biological data available for most worldwide fish stocks, making it difficult to estimate current abundance and productivity using traditional stock assessment methods. In developed countries, between 10% and 50% of fish stocks are subject to assessment, while in developing countries, this percentage often runs between 5% and 20%. The fact that SSFs mostly operate in developing countries, where there are few protective measures in place and/or insufficient enforcement of any existing restrictions, also highlights other ecological issues such as bycatch of non-target species. Due to the inherent characteristics of SSFs, such as diffuse effort, far and many landing sites, and marginalization, the bycatch rates are often difficult to estimate. In addition, the conventional fisheries assessment approaches created for large-scale fisheries do not serve as an appropriate basis for the management of SSFs because they assume a comparatively simple relationship between the productive capacity of the resource and the extractive capacity of fishing fleets. Therefore, a management strategy based on benchmarks and reference points, such as the Maximum Sustainable Yield (MSY), is meaningless in the absence of data on the fleet structure, fish abundance, fishing mortality, and regulation.

To diagnose SSFs, one should evaluate the relative importance of various opportunities, strengths, and threats by synthesizing the information already available on the fishery system. These include ecological opportunities and threats that exist within the fishery system (such as overexploitation and habitat loss), social and economic processes that exist within the system (such as an excessive fishing effort/capacity, institutional capacity for management and advice, and opportunities for livelihood diversification), as well as those that originate from the outside environment. The diagnosis sets the boundaries of what is feasible at that specific stage in the evolution of a fishery and provides a picture of the history and potential of the fishery. It is important to note that a diagnosis is not, in the conventional sense, a quick evaluation of the stock status that results in management advice.

Recently, scientific attention to SSFs and data-limited fisheries has been increasing, as evidenced by a remarkable increase in peer-reviewed papers regarding this topic in the last decade. Here, we list some examples of the studies that have analyzed the important issues affecting SSFs such as, among other things, adaptation pathways, adaptive capacity, and the impacts of climate change on SSFs [2–5], estimating the fishing effort of SSFs [6], market-based management and market weaknesses and opportunities for SSFs [7,8], reconstruction of historical fishing effort and catch per unit effort (CPUE) for SSFs [9], extinction risk, reconstructed catches and management for chondrichthyan in SSF [10], reducing the impacts of SSFs on marine megafauna [11], evaluating the ecosystem impacts of gear regulations in SSFs [12], the capacity of SSFs to provide food security [13], insights into Illegal, Unreported and Unregulated (IUU) fishing activities in SSFs [14], an assessment of the factors that influence the willingness of small-scale fishers to adopt property rights co-management options [15], an assessment of the MSY and related indicators for the main target species stocks [16], and the identification and forecast of potential fishing grounds for the main target species [17]. As the impact of climate change on SSFs is a critical issue, Salgueiro-Otero et al. [2] relied on a social–ecological network and sociodemographic data collected via face-to-face interviews with 404 small-scale commercial fishers from nine Galician communities (Spain), to empirically examine the adaptation pathways that fishers follow when they face hypothetical impacts on their fishery resources and test the role of five social-ecological network structures on fisher's stated intended responses to such scenarios. To estimate the fishing effort of SSFs, Behivoke et al. [6] monitored the movements of a sample of 31 traditional sailing fishing boats at around 45 s time intervals using small GPS trackers. A total of 306 daily tracks were recorded among five gear types (beach seine, mosquito trawl net, gillnet, handline, and speargun). To ground-truth the GPS location data, fishers' behavior was simultaneously recorded by a single onboard observer for 49 tracks. Typical, gear-specific track patterns were observed. Their findings showed that boat tracking combined with onboard observation would improve the reliability of spatial fishing effort indicators in SSFs and contribute to more efficient management. Additionally, Zeller et al. [9] reconstructed and investigated trends in the fishing effort and CPUE of SSFs in four Exclusive Economic Zones (EEZ) that constitute the Mozambique Channel, i.e., Union of Comoros, Madagascar, Mayotte, and Mozambique, from 1950 to 2016. The results indicate that the increased motorization combined with substantial growth in the overall vessel numbers were the drivers of the increasing fishing effort and decreasing CPUE and clearly suggest that continuing to increase the fishing capacity of SSF in the absence of effective and restrictive management actions may exacerbate overexploitation risk. Reducing fisheries' impacts on marine megafauna is particularly challenging in SSFs; for this reason, Booth et al. [11] presented a novel combination of methods—scenario interviews with contingent valuation (CV)—for exploring and designing locally appropriate payments for ecosystem services (PES) schemes; and apply these methods to investigate how different types of incentives might influence fisher behavior and mortality of critically endangered taxa in two case study SSF in Indonesia. In addition, in the lack of time series to estimate the predator–prey interactions (vulnerabilities) using Ecosystem models, such as Ecopath with Ecosim (EwE), Rehren et al. [12] explored available approaches for estimating the vulnerabilities to simulate the effects of a dragnet prohibition with and without reallocation of fishing effort. The results suggest that banning dragnets would be beneficial for the fishing community, judged by the increase in biomass of functional groups and fishers' profits, but not if dragnet fishers were to continue fishing in the bay by reallocating to other gears, indicated by the reduced fish biomass and fishers' profits. As for food security, Canty and Deichmann [13] evaluated long-term trends of marine SSF-landed catches in 85 Developing Economy Countries (DECs), and analyzed whether the yields of SSFs have the capacity to provide the coastal populations of DECs with a recommended annual intake (RAI) of 10.6 kg of fish per person, and how that capacity has changed over the period of 1960 to 2016. The results of the study demonstrated that landed catches of SSF alone are not a useful proxy for food security.

In the middle of all these efforts to find genuine, viable solutions to fisheries issues, it is evident that information concerning SSFs is still relatively scarce. The literature covers relatively little quantitative information on SSFs compared to large-scale fisheries. Except for the most recent efforts to gather information about SSFs through projects such as Too big to be ignored (http://toobigtoignore.net/tbti-contribution-to-ssf-knowledge/, accessed on 10 December 2022), much less has been undertaken to address the lack of comprehensive and systematic data on SSFs. Far less so is the analysis of gaps and challenges encountered in the assessment and management of SSFs, as well as an understanding of how SSFs are assessed and managed.

This Special Issue aims to provide new insights and empirical knowledge and collect original and high-quality manuscripts related to all aspects of small-scale and data-limited fisheries, such as the activity of SSF; an assessment of the effectiveness of management strategies; IUU fishing; management measures, regulations, policies, and strategies; monitoring programs; stock assessments; the sustainable development of fisheries; the sustainable exploitation of resources.

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References

1. FAO. (Ed.) *Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries in the Context of Food Security and Poverty Eradication*; FAO: Rome, Italy, 2015. Available online: https://www.fao.org/documents/card/en/c/I4356EN (accessed on 10 December 2022).
2. Salgueiro-Otero, D.; Barnes, M.L.; Ojea, E. Climate adaptation pathways and the role of social-ecological networks in small-scale fisheries. *Sci. Rep.* **2022**, *12*, 15526. [CrossRef] [PubMed]
3. Green, K.M.; Selgrath, J.C.; Frawley, T.H.; Oestreich, W.K.; Mansfield, E.J.; Urteaga, J.; Swanson, S.S.; Santana, F.N.; Green, S.J.; Crowder, L.B. How adaptive capacity shapes the Adapt, React, Cope response to climate impacts: Insights from small-scale fisheries. *Clim. Change* **2021**, *164*, 15. [CrossRef]
4. Hamilton, M.; Robinson, J.P.; Benkwitt, C.E.; Wilson, S.K.; MacNeil, M.A.; Ebrahim, A.; Graham, N.A. Climate impacts alter fisheries productivity and turnover on coral reefs. *Coral Reefs* **2022**, *41*, 921–935. [CrossRef]
5. Ilosvay, X.É.E.; Molinos, J.G.; Ojea, E. Stronger adaptive response among small-scale fishers experiencing greater climate change hazard exposure. *Commun. Earth Environ.* **2022**, *3*, 246. [CrossRef]
6. Behivoke, F.; Etienne, M.P.; Guitton, J.; Randriatsara, R.M.; Ranaivoson, E.; Léopold, M. Estimating fishing effort in small-scale fisheries using GPS tracking data and random forests. *Ecol. Indic.* **2021**, *123*, 107321. [CrossRef]
7. Nielsen, M.; Andersen, P.; Asche, F.; Ellefsen, H.; Hammarlund, C.; Hoff, A.; Kristofersson, D.M.; Nielsen, R.; Rógvi, H.; Roll, K. Can small-scale fisheries survive market-based management? *Nordic evidence. Fish Fish.* **2022**, *23*, 256–272. [CrossRef]
8. Penca, J.; Said, A.; Cavallé, M.; Pita, C.; Libralato, S. Sustainable small-scale fisheries markets in the Mediterranean: Weaknesses and opportunities. *Marit. Stud.* **2021**, *20*, 141–155. [CrossRef] [PubMed]
9. Zeller, D.; Vianna, G.M.S.; Ansell, M.; Coulter, A.; Derrick, B.; Greer, K.; Noël, S.-L.; Palomares, M.L.D.; Zhu, A.; Pauly, D. Fishing effort and associated catch per unit effort for small-scale fisheries in the Mozambique Channel region: 1950–2016. *Front. Mar. Sci.* **2021**, *8*, 707999. [CrossRef]
10. Talwar, B.S.; Anderson, B.; Avalos-Castillo, C.G.; del Pilar Blanco-Parra, M.; Briones, A.; Cardenosa, D.; Carlson, J.K.; Charvet, P.; Cotton, C.F.; Crysler, Z.; et al. Extinction risk, reconstructed catches and management of chondrichthyan fishes in the Western Central Atlantic Ocean. *Fish Fish.* **2022**, *23*, 1150–1179. [CrossRef]
11. Booth, H.; Ramdlan, M.S.; Hafizh, A.; Wongsopatty, K.; Mourato, S.; Pienkowski, T.; Adrianto, L.; Milner-Gulland, E.J. Designing locally-appropriate conservation incentives for small-scale fishers. *Biol. Conserv.* **2023**, *277*, 109821. [CrossRef]
12. Rehren, J.; Coll, M.; Jiddawi, N.; Kluger, L.C.; Omar, O.; Christensen, V.; Pennino, M.G.; Wolff, M. Evaluating ecosystem impacts of gear regulations in a data-limited fishery—Comparing approaches to estimate predator–prey interactions in Ecopath with Ecosim. *ICES J. Mar. Sci.* **2022**, *79*, 1624–1636. [CrossRef]
13. Canty, S.W.; Deichmann, J.L. Do small-scale fisheries have the capacity to provide food security to coastal populations? *Fish Fish.* **2022**, *23*, 708–718. [CrossRef]
14. Samy-Kamal, M. Insights on Illegal, Unreported and Unregulated (IUU) Fishing Activities by Egyptian Vessels in Neighbouring Countries. *Fishes* **2022**, *7*, 288. [CrossRef]
15. Kpanou, S.B.V.K.; Kpenavoun Chogou, S.; Hounnou, F.E.; Aoudji, A.K.; Lalèyè, P.A.; Dedehouanou, H.; Dogot, T. Drivers of Small-Scale Fishers' Willingness to Adopt Property Rights Co-Management in the Lake Nokoué and Porto-Novo Lagoon Complex in Southeast Benin. *Fishes* **2022**, *7*, 249. [CrossRef]
16. Andrašūnas, V.; Ivanauskas, E.; Švagždys, A.; Razinkovas-Baziukas, A. Assessment of Four Major Fish Species Stocks in the Lithuanian and Russian Parts of Curonian Lagoon (SE Baltic Sea) Using CMSY Method. *Fishes* **2022**, *7*, 9. [CrossRef]
17. Armas, E.; Arancibia, H.; Neira, S. Identification and Forecast of Potential Fishing Grounds for Anchovy (*Engraulis ringens*) in Northern Chile Using Neural Networks Modeling. *Fishes* **2022**, *7*, 204. [CrossRef]

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Article

Effects of Fishing Selectivity and Dynamics on the Performance of Catch-Based Data-Limited Assessment Models for Species with Different Life History Traits

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Abstract: The assessment of fish stocks is often limited by a lack of comprehensive data. Therefore, catch-based methods are increasingly being used because of the availability of more catch data. However, catch-based models may perform differently for species with different traits and fishing histories. In this study, we investigated the performance of catch-based models for species with different life history traits, fishing histories, and under different length selections. We compared simulated biomass with estimated stock status from three widely used catch-based models (Catch-MSY model [CMSY]; catch-only model-sampling importance resampling model [COM-SIR]; state-space catch-only model [SSCOM]) under three fishing history scenarios (constant, increasing then decreasing, and continuously increasing fishing mortality) and three length selectivity scenarios (no selectivity, preferring smaller individuals, preferring larger individuals). Our results showed that CMSY performed the best, particularly when fishing mortality remained constant. Catch-based models performed better for opportunistic species that had larger individuals selected for fishing and equilibrium species that had smaller individuals selected. However, the models tended to overestimate stock status when fishing mortality continued to increase. Therefore, caution should be exercised when applying catch-based methods to data-poor stocks with diverse life history traits, fishing history, and those sensitive to selective fishing.

Keywords: catch-based model; selective fishing; stock synthesis; data-poor fisheries; diverse life history traits

Key Contribution: This study provided the first evaluation on how fishing selectivity affects the performance of catch-based data-limited models for species with different life history traits and fishing dynamics, using simulated size-structured populations.

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1. Introduction

Assessing stock status is a critical step for sound fishery management; however, the majority of fish stocks remain unassessed because of data limitations [1]. While there is increasing attention on the sustainable use of fishery resources, only 20% of the global catch comes from assessed stocks [2]. In other words, there is insufficient information and/or capacity to assess the status of most commercially exploited species. Costello et al. (2012) [3] indicated that 64% of unassessed fisheries may be exploited stocks whose biomass (B) is below the level of the maximum sustainable yield (Bmsy), and that 18% of the unassessed stocks are considered to be collapsed (i.e., B/Bmsy < 0.2). While most of the unassessed fisheries were in developing countries and were small-scale fisheries [4], simple and cost-effective assessment methods are urgently needed to assess the stock status of these fisheries [5].

Data-limited assessment models (DLMs) have been developed to provide alternative assessment methods for species with limited data. Depending on the information required by the models, the DLMs can be broadly divided into length-based and catch-based models. Both types of methods have been used to estimate the stock status of species and to develop feasible fishery management. Examples of applying DLMs in stock assessment range from developed countries to developing countries [6–9]. Zhou et al. (2017) [7] even pointed out that DLMs can provide estimates similar to comprehensive stock assessments.

The catch-based models mainly rely on catch data to estimate stock status and other reference points for fishery management. Catch data are often the most available data for data-poor stocks, and can therefore be used as important information on stock status. For example, catch has been found to be correlated with biomass at the maximum sustainable yield (Bmsy) [3,7]. Catch data have also been used to estimate the depletion rates in population dynamics [10–12]. However, it has been argued that catch is not only correlated with population abundance, but can be affected by factors such as fishing effort [13], catchability [14,15], selectivity [16], and fishery management [14]. Therefore, these effects should be carefully considered when applying catch-based models [15].

Catch-based models may perform differently under different fishing conditions or for species with different life history traits. However, few studies have systematically analyzed the effects of life history traits and fishing mortality history on the performance of catch-based models. Hilborn et al. (2020) [4] showed that there can be complicated interactions between species biomass, fishing mortality, and catches. Carruthers et al. (2012) [17] used two catch-based models [18,19] to assess categorical stock status (e.g., developing or fully exploited), and found that the models had different classification biases for stock status under different effort trend assumptions. Free et al. (2020) [17] showed that the catch-based models had different biases under different exploitation rates and for species with different traits. However, while some studies have investigated the effects of fishing history and life history traits on the performance of catch-based models [8,17], the interactions between fishing and the life history traits of a stock have not been explored.

Selective fishing causes the distribution of the length/age composition of the catch to be skewed in a particular direction [20]. Selective fishing may occur for several reasons, including: (1) fishing gear and method—the mesh or hook targets a specific size of individuals or excludes smaller individuals; (2) fishing area and season—fishing may occur in a specific area and during the feeding or spawning ground/season, resulting in catching a specific size range of a population; (3) fishery management or the market—the government or market encourages catching large individuals. Long-term and highly selective fishing can the alter life history traits of a population [21], limit the reproductive potential of the population [22], and increase population fluctuation [23]. Given that selectivity has a profound effect on the stability of a population, it should be taken into account when estimating stock status. Catch-based models usually only consider the total weight of the catch and neglect the size/age structure of the catch. The impact of size or age selectivity on the performance of catch-based models has not been thoroughly explored.

In this study, we aim to investigate how length-selective fishing and changes in fishing mortality can affect the performance of catch-based models, considering species with different life history traits. We simulated the stock dynamics in thirty-six different scenarios, considering four types of life history strategies, three types of fishing trends, and three types of length selectivity. Our systematic investigation of the drivers of the performance of catch-based models will shed light on the application of DLMs for data-limited stock assessment.

2. Materials and Methods

2.1. Operation Model for Simulating Population Dynamics and Catch Time Series

The operation model (OM) used in this study was an age-structured model with deterministic processes considering survival rate, growth rate, maturity, fecundity, and recruitment over a simulated time period [24]. The details on the population dynamics model can be found in Appendix A of Methot and Wetzel (2013) [25]. We tested various sets of life history trait inputs to represent species with different life histories, then categorized the species into four life strategists (see details in 2.2 Simulation scenarios). Parameters of life history traits were collected from the literature for the representative species of each group (Table S1). The distribution of the variation of recruitment was assumed to be the same for all groups (a lognormal distribution with σ was between 0 and 1). We simulated a 100-year population dynamic time series with recruitment process errors. We then calculated the total biomass (By) across all ages for each year, y, as follows:

$$\mathrm{B}_y = \sum\nolimits_{a=0}^{A} w_{a,y} \times \sum\nolimits_{a=0}^{A} N_{a,y} \tag{1}$$

where A is the maximum age of the stock, $w_{a,y}$ is the mean weight of individuals at a certain age, a, in a given year, y, while $N_{a,y}$ is the number of fishes at a certain age (a) in a given year (y).

After simulating the time series of the population's biomass, we further produced the time series of the catch using the Baranov catch equation (Methot et al., 2020) [26]:

$$C_{ya} = N_{ya} \times \frac{F_y S_a}{Z_{ya}} \left(1 - e^{-Z_{ya}}\right) \tag{2}$$

where C_{ya} is the catch of an age class (a) in a year (y) N_{ya} is the biomass of the age class (a) in the year (y), F_y is the fishing mortality in the year, y, S_a is the selectivity of the age class (a), Z_{ya} is the total mortality of the age class (a) in the year (y). We determined the fishing selectivity (S) based on different selectivity scenarios (see Section 2.2 Scenarios).

The unit of the total biomass and the catch was metric tons. The operation model was conducted using the "ss3sim" package [24] in R program (version 4.0.0) for 10 iterations, then we calculated the mean and standard error of the simulated biomass for all iterations for each year.

2.2. Simulation Scenarios

In this study, we simulated population dynamics for 36 scenarios, including combinations of four life history strategies, three fishing history dynamics, and three fishing selectivity types.

Firstly, we collected life history trait information for 42 species described in [27], including length at 50% maturity, maximum length, growth coefficient, fecundity, egg size, and maximum age (Table S1). The information was used to classify the different life history trait strategies [27], as well as being used to construct the operation model for each species. We followed [27], using principal component analysis (PCA) to classify the 42 species into five strategy groups: the opportunistic strategy (OPP), equilibrium strategy (EQU), periodic strategy (PER), salmonic strategy (SAL), and intermediate strategy (INT) (Figure S1). In this study, to compare the life history strategists with more distinct traits, we excluded the intermediate strategists. Eventually, 27 species were analyzed in this study (Table S1). After constructing OM for species in each group, we compared the assessment results among the four strategy groups.

To test the effects of fishing mortality (F) dynamics, we defined three scenarios to reflect different fishing histories (Figure 1): (i) scenario F1, where fishing mortality was constant at all times; (ii) scenario F2, where fishing mortality gradually increased and then gradually decreased; (iii) scenario F3, where fishing mortality increased continuously. The mean F over all tested years (100 years) was the same for all three scenarios (Figure 1). However, the mean F was set differently for each of the life history trait groups. This is because species with different life history traits can tolerate different levels of F before being overfished. Therefore, to evaluate the performance of the data-limited assessment models for populations of different statuses (from not overfished to overfished), we need to consider the different magnitudes of fishing mortality for different life history trait groups.

Random noise ($\sigma_{f,t}$) was added from a normal distribution to the fishing mortality at each time step:

$$F_t \sim U\left(F_t, \sigma_{f,t}\right); \sigma_{f,t} = F_{mean} \times 0.1 \quad (3)$$

where F_{mean} is the mean fishing mortality for the whole fishing period, and F_t is the fishing mortality of a year, t.

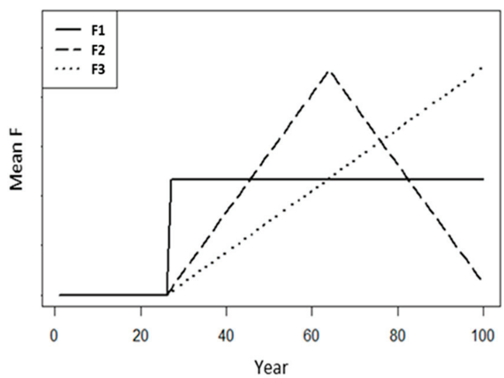

Figure 1. The simulated fishing mortality dynamics of the three F scenarios (F1, F2, and F3): F1, where fishing started in the 27th year and then remained constant (solid line); F2, where fishing started in the 27th year, increased until the 64th year, and then decreased afterwards (dashed line); F3, where fishing started in the 27th year and continuously increased (dotted line). Note that the mean F was the same for the three scenarios for the same species, but that the mean F may be different between species.

We also tested whether catches from selective fisheries affect the performance of catch-based models. Since selective fishing is typically based on length rather than age in practice, we simulated length selection in this study. In other words, we assumed that a selection of length was equivalent to a selection of age for fishes [28]. We simulated three different length selectivity scenarios (S). The first scenario, S0, represented a non-selective fishery, where the chance of being caught was constant across all length classes. The second scenario, S1, simulated a fishery where smaller individuals had a greater probability of being caught. The third scenario, S2, represented a fishery where larger individuals had a greater probability of being caught (see an example in Figure 2). We used the method of double normal distribution with plateau to determine the probability of selecting individuals in each length class. To determine the peaks of selectivity, we defined the length with the highest selection probability for S1 ($S1_{top}$) and S2 ($S2_{top}$) for each species using the following rules: $S1_{top} = L_{min} + (L_{max} - L_{min})/9*3$, $S2_{top} = L_{min} + (L_{max} - L_{min})/9*6$, where L_{min} is the minimum length and L_{max} is the maximum length of a species (Table S1). The L_{min} is the length at t_0 of the von Bertalanffy growth function (VBGF) of the species, collected from Fishbase. The selectivity was between 0 and 1, and the mean of S across all lengths was set as identical for all three scenarios of each species.

Overall, there were nine scenarios of fishing conditions (3 F-scenarios $\times$ 3 S-scenarios) for twenty-seven species belonging to four life history strategy groups.

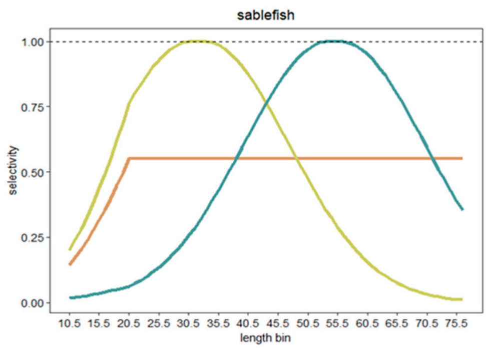

Figure 2. An example of the simulated three length selectivity scenarios (S0, S1, S2) for sablefish (*Anoplopoma fimbria*). S0 represents a non-selective fishery (orange); S1 is where smaller individuals had a greater probability of being caught (yellow); S2 is where larger individuals had a greater probability of being caught (green).

2.3. Catch-Based Models

For each species under each scenario, three catch-based assessment models were applied to the simulated catch data to evaluate model performance. The catch-based models used in this study include the Catch-MSY model [CMSY] [10], catch-only model-sampling importance resampling model [COM-SIR] [11], and state-space catch-only model [SSCOM] [12]. The parameters used in the catch-based models for each species were listed in Table S1. The analyses were conducted by the "datalimited" package [29] in R program (version 4.0.0).

The CMSY [10] required six types of input data, including catch time-series, resilience (r), natural mortality (M), carrying capacity (K), and the range of the depletion rate for the first and the final year. We determined that the priors of r and K were from uniform distributions, and we used the Bernoulli distribution as the likelihood function for viable r–K pairs. To obtain enough pairs of the viable r–K combinations (those with which the population would not collapse or be over the carrying capacity, as defined in [10]), we assumed the maximum value of K as 100 times the maximum catch and the minimum value of r as 0.05. The depletion rate was a default value that was derived from the catch at year 0 and the last year divided by the maximum catch (following [10]). The same prior settings of the range of r and K were also used for the COM-SIR and SSCOM, but the prior distributions were assumed to be normal and lognormal in the COM-SIR and SSCOM, respectively.

For the COM-SIR [11], we assumed the harvest rate as a logistic function that was controlled by the parameters a (the bioeconomic equilibrium as a proportion of K) and x (the intrinsic rate of effort change). We defined the prior probability distribution for each of the parameters as a~U(0, 1) and x~U(0.000001, 1).

For the SSCOM [12], we defined the prior distribution for the model parameters also as uniform distributions: a~U(0, 2) and x~U(0.01, 0.5). Since the SSCOM considers the stochasticity of the population dynamics, effort dynamics, and catch efficiency using a Bayesian state-space framework, we also defined the process errors of effort, biomass, and catchability as $\sigma_E, \sigma_P, \sigma_Q$~U (0.01, 1).

2.4. Evaluating the Performance of the Catch-Based Models

In order to examine the performance of catch-based models in estimating stock status under different fishing scenarios, we calculated the relative error (RE) between the model-estimated and the simulated stock biomass (B) proportional to the Bmsy (BBmsy):

$$\mathrm{RE} = \frac{\hat{\theta} - \theta}{\theta} \tag{4}$$

where $\hat{\theta}$ is the estimated value from the catch-based model and θ is the BBmsy value calculated from the operating model.

3. Results

3.1. Simulation Outputs by Life History Traits and Fishing Strategy

The trends in simulated biomass varied among the F scenarios, life history trait strategies, and S scenarios (Figure S2). In general, species biomass remained stable at constant fishing mortality (F1) after a drop in the first year of fishing. In contrast, for scenario F2, species biomass decreased with increasing fishing mortality and increased with a time lag as fishing mortality decreased. In the scenario where the fishing mortality increased continuously (F3), the species biomass continuously decreased.

The biomass of species with different life history trait strategies had different responses to length selectivity (Figure S2). For the OPP strategists, fishing that favored larger individuals (S2) generally resulted in higher species biomass. In contrast, for the EQU strategists, a preference for smaller individuals (S1) resulted in a greater species biomass. For the SAL and PER strategists, the biomass was not affected by length selectivity.

Catch changed positively with the level of fishing mortality, but often decreased when the fishing mortality reached a certain level and did not increase again even when F decreased in the F2 scenario (Figure 3). For all life history trait strategies, greater selectivity for larger individuals (S2) resulted in higher catches than the scenario that favored smaller individuals (S1) (Figure 3).

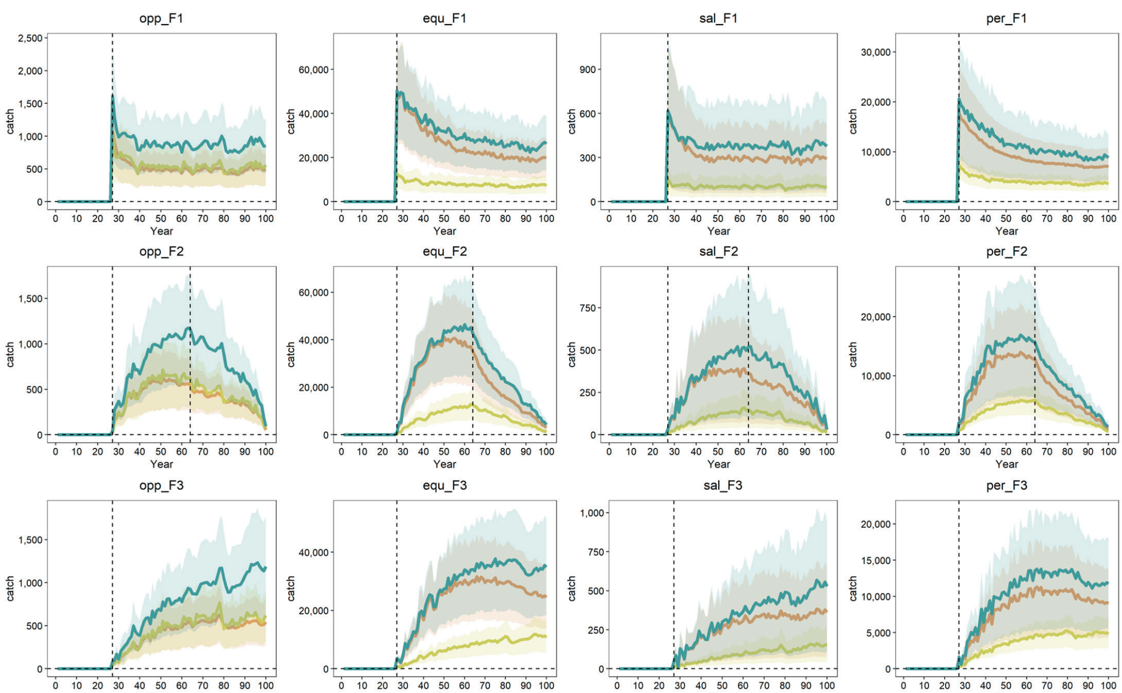

Figure 3. The simulated catch time series (unit: Mt) of four life history strategy groups (OPP, EQU, SAL, and PER) under three F scenarios (F1, upper panel; F2, middle panel; F3, bottom panel) and three S scenarios (S0, orange; S1, yellow; S2, green). The line is the mean catch of all species of each life history strategy, with the standard errors as the shaded areas.

3.2. Evaluating the Performance of the Catch-Based Models

The relative errors between the catch-based model and the operational model outputs were the smallest for CMSY (Figure 4). In addition, our results showed that the catch-based models had different performances under different fishing selectivity, especially for the OPP and EQU species (Figure 4). We found that for the OPP species, catch-based models performed better (i.e., with a lower RE) under scenario S2, where fishing targeted the larger individuals (Figure 4a). In contrast, for the EQU species, the catch-based models performed better when fishing targeted the smaller individuals (scenario S1) (Figure 4b). REs were generally not significantly different between the S scenarios for the SAL and PER species (Figure 4c,d).

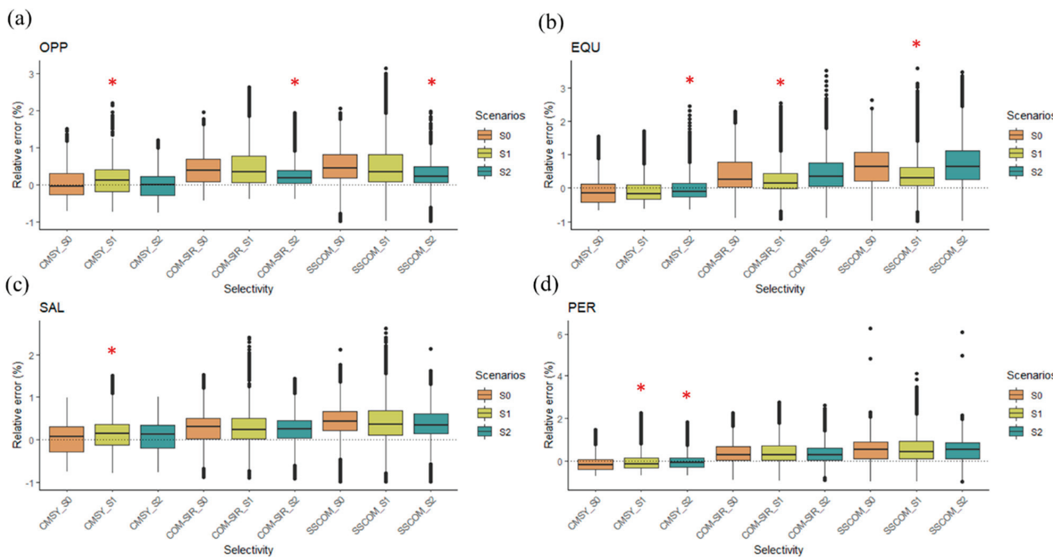

Figure 4. The boxplots show the relative errors between the estimates of the catch-based models and the simulated true BBmsy for (**a**) oppertunistic species (OPP), (**b**) equalibrium species (EQU), (**c**) salmonic species (SAL), and (**d**) periodic speices (PER). The red star indicates the significant difference between the scenario and S0.

When comparing them among scenarios with different fishing histories, our results showed that the catch-based models generally performed the best under the F2 scenario (increasing then decreasing fishing). However, the model estimates lagged in reflecting biomass recovery when the fishing mortality was decreasing in the later part of F2 (Figure 5). For the constant fishing mortality scenario (F1), the catch-based models performed better for the OPP and SAL species compared to the EQU and PER species (Figure 5). In addition, the REs of the CMSY remained stable throughout the time series for all life history strategies, whereas the COM-SIR and SSCOM produced increasing REs for the OPP and SAL strategies (Figure 5). For the continuously increasing fishing scenario (F3), the estimated BBmsy values were increasingly over-estimated by the three catch-based models, while the biomass actually decreased, resulting in an increasing RE over time (Figure 5). Overall, the CMSY performed the best among the three models for all F scenarios.

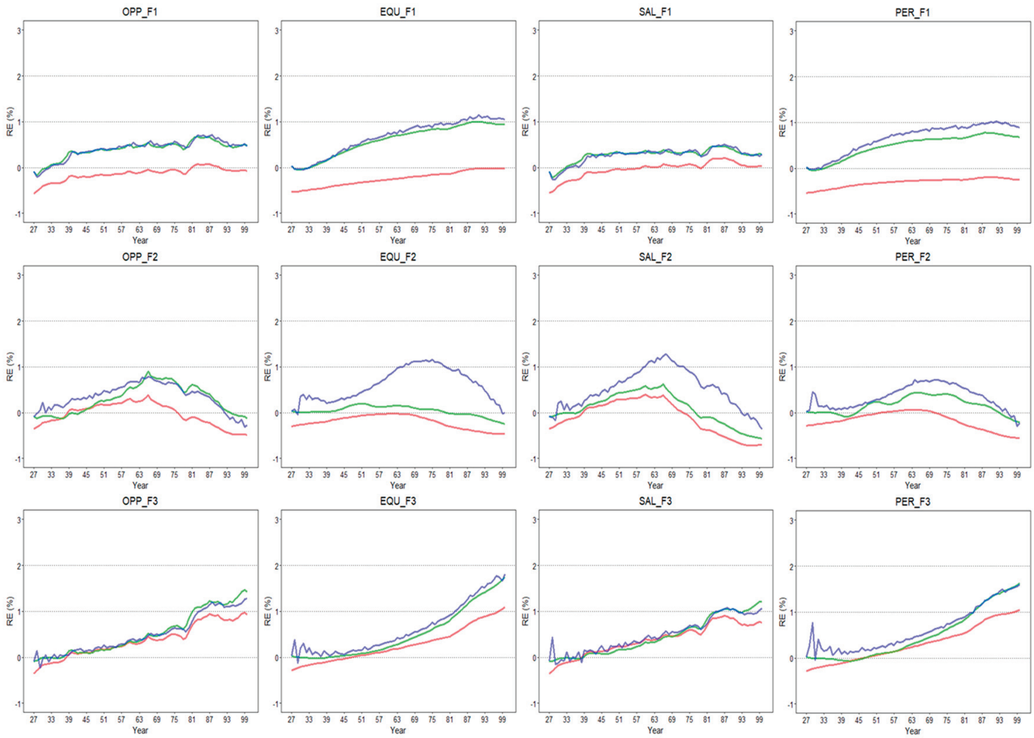

Figure 5. The relative errors from the 27th year to 100th year of the four life history strategy groups (OPP, EQU, SAL, and PER) under the three F-scenarios (F1, F2, and F3). Red represents the REs estimated from the CMSY; green represents theREs estimated from the COM-SIR; blue represents the REs estimated from the SSCOM.

4. Discussion

We found that the CMSY performed the best for species with different life history traits and under different fishing scenarios among the three commonly used catch-based assessment methods. Such results support previous studies that suggest that CMSY estimates deviate less from the true or assessed biomass status for the comprehensive models than for many other data-limited models [8,30,31]. We also found that catch-based models performed differently with respect to fishing selectivity, with different responses for species with different life history traits. This is the first study examining the effects of selective fishing on the performance of catch-based models. The catch-based models generally had more stable performance at constant fishing mortality, as these models struggled to reflect biomass recovery and overestimated stock status especially when fishing mortality continued to increase. Our results have important implications for the use of catch-based models in the assessment of data-poor fisheries, in particular highlighting the importance of accounting for fishing selectivity and life history traits of species.

Our research indicates that the selective fishing of opportunistic and equilibrium species can impact the performance of catch-based models. This suggests that when using catch-based DLMs, the size and age structure of the stock should be taken into consideration. We found that the three catch-based models performed better when fishing favored larger individuals for opportunistic species and when it favored smaller individuals for equilibrium species. For opportunistic strategists, the extraction of larger individuals would be less damaging to the size structure of the population, given their high resilience [32]. The population under scenario S2 could therefore have had a higher biomass, which would result in a smaller difference in the optimistic estimate than that of catch-based models. In contrast, for equilibrium strategists, the selection of larger individuals would have a greater impact on the stability of the population biomass [23]. The stock status estimated by catch-based models would therefore be overestimated.

Of the three catch-based models tested, the CMSY was found to perform best and to be stable, although all three models showed increasing REs as the fishing mortality continued to increase. Our finding that the CMSY had the best performance was consistent with that of previous studies comparing the CMSY with many other data-limited models [8,30,31]. When the fishing mortality remained constant (scenario F1), the REs of the CMSY remained the smallest and the most stable over the entire simulation period for all life history strategies. In contrast, the COM-SIR and SSCOM produced increasing REs for the OPP and SAL strategies. The stable errors between the years provided opportunities to apply these methods in the stock assessment, as the bias could be easily corrected and interannual comparisons could be made, as also demonstrated by Pons et al. (2020) [8]. In scenario F2 (where fishing mortality decreased after increasing), we found that only the CMSY could detect biomass recovery for all life-history strategies. Our results support the assertion that for the COM-SIR and SSCOM, the effort dynamics should be modelled separately before and after the fishery management is involved [12]. In F3 (which had increasing fishing mortality), the REs of the three catch-based models increased over time, indicating that the catch-based models failed to capture the increasingly exploited state of the stocks. However, we note that a prior study discovered that all catch-based models currently in use encountered challenges when attempting to produce reliable outcomes that aligned with traditional stock assessment methods, despite having access to extensive data [17]. This indicates that catch-based models should still be used with caution.

The differences in the performance of catch-based models for different types of life history traits have been identified in our results. The results of our study support that the life history traits of a species not only directly affect the predictability of catch-based models [17,33], but that the traits also correspond to fishing selectivity in terms of the effect on the performance of DLMs. Given the effects of life history traits, the choice of priors for traits relevant to population growth rate and life span (e.g., r and K) would be critical in the application of these models [10–12]. Martell and Froese (2013) [10] therefore suggested that the best available knowledge should be used to decide the priors for r and K, including a consideration of expert opinion and the use of meta-analysis to estimate these values [34].

We note that the assessment results of some catch-based models should be interpreted with caution, as the assessed biomass may be highly correlated with the catch but not with the true biomass dynamics. For example, in our results, the BBmsy estimated by the SSCOM was more similar to the annual variation of catch than to the biomass. Thorson et al. (2013) [12] also pointed out that the SSCOM is highly dependent on the catch time series. Considering that catch data can be affected by many factors in addition to stock biomass, such as market shifts and changes in fishing regulations, the trend of the catch time series could deviate from the biomass [35,36]. We therefore recommend that catch-based models should be applied more carefully to avoid mis-estimating stock status [37].

The development of data-limited methods should not prevent the government from continuously collecting information for comprehensive assessments. Because catch-based models are sensitive to fishing history and strategy [30], as our results showed, we recommend that the history of changes in fishing efforts are considered based on expert opinion or the literature when applying catch-based models. In addition, if the age/length composition of the population is sensitive to length-selective fishing, we should consider the selectivity when applying catch-based models. In a data-limited situation, stock assessment should ideally be carried out under separate fishery management and regulatory phases. While the data-limited models could provide a rapid assessment for management decisions, it should be noted that our use of these methods involved simple hypotheses and often provided biased estimates of fishery status [17]. It is recommended that the management bodies continue to monitor stock status to implement comprehensive fishery management under the regulation of the harvest control rule (HCR) and management strategy evaluation (MSE) framework [31,38].

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/fishes8030130/s1, Table S1, the life-history parameters used for simulating the population dynamics of the 27 species in the four life history strategy groups [27,39–73]; Figure S1, the results of the cluster analysis of the 42 fish species in King & McFarlane (2003); Figure S2, the time series of the simulated B/Bmsy of the four life history strategists (OPP, EQU, SAL, and PER) in the three F scenarios (F1, F2, and F3) and three S scenarios (S0, S1, and S2).

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References

1. Pilling, G.; Apostolaki, P.; Science, P.F. Assessment and Management of Data-Poor Fisheries. In *Advances in Fisheries Science: 50 Years on From Beverton and Holt*; Payne, A.I.L., Cotter, J., Potter, T., Eds.; John Wiley & Sons, Ltd.: Hoboken, NJ, USA, 2009.
2. Worm, B.; Barbier, E.B.; Beaumont, N.; Duffy, J.E.; Folke, C.; Halpern, B.S.; Jackson, J.B.C.; Lotze, H.K.; Micheli, F.; Palumbi, S.R.; et al. Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science* **2006**, *314*, 787–790. [CrossRef] [PubMed]
3. Costello, C.; Ovando, D.; Hilborn, R.; Gaines, S.D.; Deschenes, O.; Lester, S.E. Status and Solutions for the World's Unassessed Fisheries. *Science* **2012**, *338*, 517–521. [CrossRef] [PubMed]
4. Hilborn, R.; Amoroso, R.O.; Anderson, C.M.; Baum, J.K.; Branch, T.A.; Costello, C.; De Moor, C.L.; Faraj, A.; Hively, D.; Jensen, O.P.; et al. Effective Fisheries Management Instrumental in Improving Fish Stock Status. *Proc. Natl. Acad. Sci. USA* **2020**, *117*, 2218–2224. [CrossRef] [PubMed]
5. Pita, C.; Villasante, S.; Pascual-Fernández, J.J. Managing Small-Scale Fisheries under Data Poor Scenarios: Lessons from around the World. *Mar. Policy* **2019**, *101*, 154–157. [CrossRef]
6. Sagarese, S.R.; Rios, A.B.; Cass-Calay, S.L.; Cummings, N.J.; Bryan, M.D.; Stevens, M.H.; Harford, W.J.; McCarthy, K.J.; Matter, V.M. Working towards a Framework for Stock Evaluations in Data-Limited Fisheries. *N. Am. J. Fish. Manag.* **2018**, *38*, 507–537. [CrossRef]
7. Zhou, S.; Punt, A.E.; Ye, Y.; Ellis, N.; Dichmont, C.M.; Haddon, M.; Smith, D.C.; Smith, A.D. Estimating Stock Depletion Level from Patterns of Catch History. *Fish Fish.* **2017**, *18*, 742–751. [CrossRef]
8. Pons, M.; Cope, J.M.; Kell, L.T. Comparing Performance of Catch-Based and Length-Based Stock Assessment Methods in Data-Limited Fisheries. *Can. J. Fish. Aquat. Sci.* **2020**, *77*, 1026–1037. [CrossRef]
9. van Gemert, R.; Koemle, D.; Winkler, H.; Arlinghaus, R. Data-Poor Stock Assessment of Fish Stocks Co-Exploited by Commercial and Recreational Fisheries: Applications to Pike Esox Lucius in the Western Baltic Sea. *Fish. Manag. Ecol.* **2022**, *29*, 16–28. [CrossRef]
10. Martell, S.; Froese, R. A Simple Method for Estimating MSY from Catch and Resilience. *Fish Fish.* **2013**, *14*, 504–514. [CrossRef]
11. Vasconcellos, M.; Cochrane, K. Overview of World Status of Data-Limited Fisheries: Inferences from Landings Statistics. In *Fisheries Assessment and Management in Data-Limited Situations*; Krus, G.H., Gall, V.F., Hay, D.E., Perry, R.I., Peterman, R.M., Shirley, T.C., Spencer, P.D., Wilson, B., Woodby, D., Eds.; Alaska Sea Grant, Univ.: Fairbanks, AK, USA, 2005; pp. 1–20.
12. Thorson, J.T.; Minto, C.; Minte-Vera, C.V.; Kleisner, K.M.; Longo, C. A New Role for Effort Dynamics in the Theory of Harvested Populations and Data-Poor Stock Assessment. *Can. J. Fish. Aquat. Sci.* **2013**, *70*, 1829–1844. [CrossRef]
13. Harley, S.J.; Myers, R.A.; Dunn, A. Is Catch-per-Unit-Effort Proportional to Abundance? *Can. J. Fish. Aquat. Sci.* **2001**, *58*, 1760–1772. [CrossRef]
14. Macusi, E.D.; Morales, I.D.G.; Macusi, E.S.; Pancho, A.; Digal, L.N. Impact of Closed Fishing Season on Supply, Catch, Price and the Fisheries Market Chain. *Mar. Policy* **2022**, *138*, 105008. [CrossRef]

15. Langlois, T.J.; Newman, S.J.; Cappo, M.; Harvey, E.S.; Rome, B.M.; Skepper, C.L.; Wakefield, C.B. Length Selectivity of Commercial Fish Traps Assessed from in Situ Comparisons with Stereo-Video: Is There Evidence of Sampling Bias? *Fish. Res.* **2015**, *161*, 145–155. [CrossRef]
16. Carruthers, T.R.; Walters, C.J.; McAllister, M.K. Evaluating Methods That Classify Fisheries Stock Status Using Only Fisheries Catch Data. *Fish. Res.* **2012**, *119–120*, 66–79. [CrossRef]
17. Free, C.M.; Jensen, O.P.; Anderson, S.C.; Gutierrez, N.L.; Kleisner, K.M.; Longo, C.; Minto, C.; Osio, G.C.; Walsh, J.C. Blood from a Stone: Performance of Catch-Only Methods in Estimating Stock Biomass Status. *Fish. Res.* **2020**, *223*, 105452. [CrossRef]
18. Kleisner, K.; Pauly, D. Stock-Status Plots for Fisheries for Regional Seas. In *The State of Biodiversity and Fisheries in Regional Seas, Fisheries Centre Research Reports 19(3)*; Christensen, V., Lai, S., Palomares, M.L.D., Zeller, D., Pauly, D., Eds.; Fisheries Centre, University of British Columbia: Vancouver, BC, Canada, 2011.
19. Froese, R.; Kesner-Reyes, K. *Impact of Fishing on the Abundance of Marine Species*; ICES: Copenhagen, Denmark, 2002.
20. Piet, G.J.; Rijnsdorp, A.D. Changes in the Demersal Fish Assemblage in the South-Eastern North Sea Following the Establishment of a Protected Area ("Plaice Box"). *ICES J. Mar. Sci.* **1998**, *55*, 420–429. [CrossRef]
21. Fenberg, P.B.; Roy, K. Ecological and Evolutionary Consequences of Size-Selective Harvesting: How Much Do We Know? *Mol. Ecol.* **2008**, *17*, 209–220. [CrossRef]
22. Birkeland, C.; Dayton, P.K. The Importance in Fishery Management of Leaving the Big Ones. *Trends Ecol. Evol.* **2005**, *20*, 356–358. [CrossRef]
23. Hsieh, C.H.; Reiss, C.S.; Hunter, J.R.; Beddington, J.R.; May, R.M.; Sugihara, G. Fishing Elevates Variability in the Abundance of Exploited Species. *Nature* **2006**, *443*, 859–862. [CrossRef]
24. Anderson, S.C.; Monnahan, C.C.; Johnson, K.F.; Ono, K.; Valero, J.L. Ss3sim: An R Package for Fisheries Stock Assessment Simulation with Stock Synthesis. *PLoS ONE* **2014**, *9*, e92725. [CrossRef]
25. Methot, R.D.; Wetzel, C.R. Stock Synthesis: A Biological and Statistical Framework for Fish Stock Assessment and Fishery Management. *Fish. Res.* **2013**, *142*, 86–99. [CrossRef]
26. Methot, R.D., Jr.; Wetzel, C.R.; Taylor, I.G.; Doering, K. *Stock Synthesis User Manual Version 3.30.15. U.S. Department of Commerce, NOAA Processed Report NMFS-NWFSC-PR-2020-05*; Northwest Fisheries Science Center: Seattle, WA, USA, 2020.
27. King, J.R.; McFarlane, G.A. Marine Fish Life History Strategies: Applications to Fishery Management. *Fish. Manag. Ecol.* **2003**, *10*, 249–264. [CrossRef]
28. Ricker, W.E. Handbook of Computations for Biological Statistics of Fish Populations. *Bull. Fish. Res. Board Can.* **1958**, *119*, 1–300.
29. Anderson, S.C.; Afflerbach, J.; Cooper, A.B.; Dickey-Collas, M.; Jensen, O.P.; Kleisner, K.M.; Longo, C.; Osio, G.C.; Ovando, D.; Minte-Vera, C.; et al. Datalimited: Stock Assessment Methods for Data-Limited Fisheries. R Package Version 0.1.0. 2016. Available online: https://github.com/datalimited/datalimited (accessed on 17 February 2023).
30. Rosenberg, A.A.; Fogarty, M.J.; Cooper, A.B.; Dickey-Collas, M.; Fulton, E.A.; Gutiérrez, N.L.; Hyde, K.J.W.; Kleisner, K.M.; Kristiansen, T.; Longo, C.; et al. Developing New Approaches to Global Stock Status Assessment and Fishery Production Potential of the Seas. *FAO Fish. Aquac. Circ.* **2014**, *1086*, 175.
31. Walsh, J.C.; Minto, C.; Jardim, E.; Anderson, S.C.; Jensen, O.P.; Afflerbach, J.; Dickey-Collas, M.; Kleisner, K.M.; Longo, C.; Osio, G.C.; et al. Trade-Offs for Data-Limited Fisheries When Using Harvest Strategies Based on Catch-Only Models. *Fish Fish.* **2018**, *19*, 1130–1146. [CrossRef]
32. Juan-Jordá, M.J.; Mosqueira, I.; Freire, J.; Dulvy, N.K. Population Declines of Tuna and Relatives Depend on Their Speed of Life. *Proc. R. Soc. B Biol. Sci.* **2015**, *282*, 20150322. [CrossRef]
33. Rosenberg, A.A.; Kleisner, K.M.; Afflerbach, J.; Anderson, S.C.; Dickey-Collas, M.; Cooper, A.B.; Fogarty, M.J.; Fulton, E.A.; Gutiérrez, N.L.; Hyde, K.J.W.; et al. Applying a New Ensemble Approach to Estimating Stock Status of Marine Fisheries around the World. *Conserv. Lett.* **2018**, *11*, e12363. [CrossRef]
34. Punt, A.E.; Hilborn, R. *Bayesian Stock Assessment Methods in Fisheries-User's Manual*; Food and Agriculture Organization: Rome, Italy, 2001.
35. Branch, T.A.; Jensen, O.P.; Ricard, D.; Ye, Y.; Hilborn, R. Contrasting Global Trends in Marine Fishery Status Obtained from Catches and from Stock Assessments. *Conserv. Biol.* **2011**, *25*, 777–786. [CrossRef]
36. Froese, R.; Zeller, D.; Kleisner, K.; Pauly, D. What Catch Data Can Tell Us about the Status of Global Fisheries. *Mar. Biol.* **2012**, *159*, 1283–1292. [CrossRef]
37. Hilborn, R.; Branch, T.A. Fisheries: Does Catch Reflect Abundance? *Nature* **2013**, *494*, 303–306.
38. Hordyk, A.R.; Loneragan, N.R.; Prince, J.D. An Evaluation of an Iterative Harvest Strategy for Data-Poor Fisheries Using the Length-Based Spawning Potential Ratio Assessment Methodology. *Fish. Res.* **2015**, *171*, 20–32. [CrossRef]
39. Sassa, C.; Ohshimo, S.; Tanaka, H.; Tsukamoto, Y. Reproductive Biology of *Benthosema Pterotum* (Teleostei: Myctophidae) in the Shelf Region of the East China Sea. *J. Mar. Biol. Assoc. U. K.* **2014**, *94*, 423–433. [CrossRef]
40. Garcia Seoane, E. *Ecology of the Glacier Lanternfish Benthosema Glaciale (Reinhardt,1837) with Focus on the Flemish Cap*; University of Vigo: Vigo, Spain, 2013.
41. Brand, E.J.; Kaplan, I.C.; Harvey, C.J.; Levin, P.S.; Fulton, E.A.; Hermann, A.J.; Field, J.C. A Spatially Explicit Ecosystem Model of the California Current's Food Web and Oceanography. *NOAA Tech. Memo. NMFS-NWFSC* **2007**, *84*, 145.
42. Liang, Y.; Sui, X.; Chen, Y.; Jia, Y.; He, D. Life History Traits of the Chinese Minnow *Rhynchocypris Oxycephalus* in the Upper Branch of Yangtze River, China. *Zool. Stud.* **2014**, *53*, 1–10. [CrossRef]

43. Hay, D.E.; Mccarter, P.B. Status of the Eulachon (*Thaleichthys pacificus*) in Canada. *Can. Stock Assess. Res. Doc.* **2000**, *92*.
44. Nelson, G.A.; Ross, M.R. Biology and Population Changes of Northern Sand Lance (*Ammodytes dubius*) from the Gulf of Maine to the Middle Atlantic Bight. *J. Northwest Atl. Fish. Sci.* **1991**, *11*, 11–27. [CrossRef]
45. Griffith, D.A.; Lo, N.C.H.; Jolla, L.; Griffith, D.A. Spawning Biomass of Pacific Sardine (*Sardinops sagax*), from 1994–2004 off California. *CalCOFI Rep* **2005**, *46*.
46. Galindo-Cortes, G.; De Anda-Montañez, J.A.; Arreguín-Sánchez, F.; Salas, S.; Balart, E.F. How Do Environmental Factors Affect the Stock-Recruitment Relationship? The Case of the Pacific Sardine (*Sardinops sagax*) of the Northeastern Pacific Ocean. *Fish. Res.* **2010**, *102*, 173–183. [CrossRef]
47. Haist, V.; Stocker, M. Growth and Maturation of Pacific Herring (*Clupea harengus pallasi*) in the Strait of Georgia. *Can. J. Fish. Aquat. Sci.* **1985**, *42*, 138–146. [CrossRef]
48. Burd, A.C.; Howlett, G.J. Fecundity Studies on North Sea Herring. *ICES J. Mar. Sci.* **1974**, *35*, 107–120. [CrossRef]
49. Nash, R.D.M.; Dickey-Collas, M.; Kell, L.T. Stock and Recruitment in North Sea Herring (*Clupea harengus*); Compensation and Depensation in the Population Dynamics. *Fish. Res.* **2009**, *95*, 88–97. [CrossRef]
50. Holden, M.J. The Fecundity of *Raja clavata* in British Waters. *ICES J. Mar. Sci.* **1975**, *36*, 110–118. [CrossRef]
51. Ebert, D.A.; Smith, W.D.; Cailliet, G.M. Reproductive Biology of Two Commercially Exploited Skates, *Raja binoculata* and *R. rhina*, in the Western Gulf of Alaska. *Fish. Res.* **2008**, *94*, 48–57. [CrossRef]
52. Drew, M.; White, W.T.; Dharmadi; Harry, A.V.; Huveneers, C. Age, Growth and Maturity of the Pelagic thresher *Alopias pelagicus* and the Scalloped hammerhead *Sphyrna lewini*. *J. Fish Biol.* **2015**, *86*, 333–354. [CrossRef] [PubMed]
53. Hsu, H.; Joung, S.J.; Lyu, G.T.; Liu, K.M.; Huang, C.C. Age and Growth of the Blue Shark, *Prionace glauca*, in the Northwest Pacific. In Proceedings of the Shark Working Group Workshop ISC/11/SHARKWG-2/, La Jolla, CA, USA, 28 November–6 December 2011; p. 22.
54. King, J.R.; McPhie, R.P. Preliminary Age, Growth and Maturity Estimates of Spotted Ratfish (*Hydrolagus colliei*) in British Columbia. *Deep. Res. Part II Top. Stud. Oceanogr.* **2015**, *115*, 55–63. [CrossRef]
55. Dutton, J.; Gioia, K.A. Fecundity and Embryonic Development of Spiny Dogfish in the Northwest Atlantic Ocean. *Trans. Am. Fish. Soc.* **2019**, *148*, 48–57. [CrossRef]
56. Kwain, W.H. Spawning Behavior and Early Life History of Pink Salmon (*Oncorhynchus gorbuscha*) in the Great Lakes (Lake superior). *Can. J. Fish. Aquat. Sci.* **1982**, *39*, 1353–1360. [CrossRef]
57. Godbout, L.; Fu, C.H.; Irvine, J.R. Evaluation of Chum Salmon Fishery Performance Using Ricker and Beverton-Holt Stock Recruitment Approaches in a Bayesian Framework. *Reconciling Fish. with Conserv. Vols I Ii* **2008**, *49*, 1417–1433.
58. Beacham, T.D. Fecundity of Coho Salmon (*Oncorhynchus kisutch*) and Chum Salmon (*Oncorhynchus keta*) in the Northeast Pacific Ocean. *Can. J. Zool.* **1982**, *60*, 1463–1469. [CrossRef]
59. Hartman, W.L.; Conkle, C.Y. Fecundity of Red Salmon at Brooks and Karluk Lakes, Alaska. *Fish. Bull. Fish Wildl. Serv.* **1960**, *61*, 53–60.
60. Kerns, J.A.; Rogers, M.W.; Bunnell, D.B.; Claramunt, R.M.; Collingsworth, P.D. Comparing Life History Characteristics of Lake Michigan's Naturalized and Stocked Chinook Salmon. *N. Am. J. Fish. Manag.* **2016**, *36*, 1106–1118. [CrossRef]
61. Echeverria, T.W. Thirty-Four Species of California Rockfishes: Maturity and Seasonality of Reproduction. *Fish. Bull.* **1988**, *85*, 229–250.
62. Haigh, R.; Starr, P. A Review of Yellowmouth Rockfish *Sebastes reedi* along the Pacific Coast of Canada: Biology, Distribution, and Abundance Trends. *CSAS Res. Doc.* **2008**, *3848*, 105.
63. Stanley, R.D.; Kronlund, A.R. Life History Characteristics for Silvergray Rockfish (*Sebastes brevispinis*) in British Columbia Waters and the Implications for Stock Assessment and Management. *Fish. Bull.* **2005**, *103*, 670–684.
64. Conrath, C.L.; Knoth, B. Reproductive Biology of Pacific Ocean Perch in the Gulf of Alaska. *Mar. Coast. Fish.* **2013**, *5*, 21–27. [CrossRef]
65. Love, M.S.; Morris, P.; McCrae, M.; Colins, R. Life History Aspects of 19 Rockfish Species (Scorpaenidae: Sebastes) from the Southern California Bight. *NOAA Tech. Rep. NMFS 87* **1990**.
66. Beyer, S.G.; Sogard, S.M.; Harvey, C.J.; Field, J.C. Variability in Rockfish (*Sebastes* spp.) Fecundity: Species Contrasts, Maternal Size Effects, and Spatial Differences. *Environ. Biol. Fishes* **2015**, *98*, 81–100. [CrossRef]
67. Schmitt, C.; St-Pierre, G. *Evaluation of Two Methods to Determine Maturity of Pacific Halibut*; International Pacific Halibut Commission: Seattle, WA, USA, 1997; pp. 1–24.
68. Haug, T.; Gulliksen, B. Fecundity and Oocyte Sizes in Ovaries of Female Atlantic Halibut, *Hippoglossus hippoglossus* (L.). *Sarsia* **1988**, *73*, 259–261. [CrossRef]
69. Head, M.A.; Keller, A.A.; Bradburn, M. Maturity and Growth of Sablefish, *Anoplopoma fimbria*, along the U.S. West Coast. *Fish. Res.* **2014**, *159*, 56–67. [CrossRef]
70. Hunter, J.R.; Macewicz, B.J.; Kimbrell, C.A. Fecundity and Other Aspects of the Reproduction of Sablefish, *Anoplopoma fimbria*, in Central California Waters. *CalCOFI Rep* **1989**, *30*, 61–72.
71. Schirripa, M.J.; Goodyear, C.P.; Methot, R.M. Testing Different Methods of Incorporating Climate Data into the Assessment of US West Coast Sablefish. *ICES J. Mar. Sci.* **2009**, *66*, 1605–1613. [CrossRef]

72. Pearson, K.E.; Gunderson, D.R. Reproductive Biology and Ecology of Shortspine Thornyhead Rockfish, *Sebastolobus alascanus*, and Longspine Thornyhead Rockfish, *S. altivelis*, from the Northeastern Pacific Ocean. *Environ. Biol. Fishes* **2003**, *67*, 117–136. [CrossRef]
73. Cooper, D.W.; Pearson, K.E.; Gunderson, D.R. Fecundity of Shortspine Thornyhead (*Sebastolobus alascanus*) and Longspine Thornyhead (*S. altivelis*) (Scorpaenidae) from the Northeastern Pacific Ocean, Determined by Stereological and Gravimetric Techniques. *Fish. Bull.* **2005**, *103*, 15–22.

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Socio-Ecological Overview of the Greater Amberjack Fishery in the Balearic Islands

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Abstract: This study provides the first socio-ecological overview of the *Seriola dumerili* fishery occurring in the Balearic Islands. This pelagic top-predator is among the five most valued fish resources of the Balearic community. Despite its ecological importance and potential vulnerability to aggregation fishing, few studies address the ecology of this large Carangidae species. Shining a light on its ecology is vital to ensure adequate species conservation and the sustainable and effective management of the fishery. Historical catches from 1950–1999, alongside detailed landing data for the last 21 years, were analysed to identify potential patterns in ecological and socio-economic factors. Significant inter-annual variability among the years was found in historical catches of greater amberjack, while catches and mean prices of the different size categories revealed significant results between seasons and months, respectively. Additionally, the purse seine fleet accounted for the highest percentage of *S. dumerili* catches. CPUE did not appear to change greatly between months and years after the annual 8-month fishing ban imposed in 2011 and therefore a re-evaluation of the closure was intended. Overall, this study suggests seasonality influences the *S. dumerili* fishery in the Balearic Islands, within which ecological influences show a higher regulating power than socio-economic factors.

Keywords: *Seriola dumerili*; social-ecological systems; mediterranean; historical catches; fisheries dynamics

Key Contribution: The greater amberjack fishery in the Balearic Islands is affected by both socio-economic and ecological factors, although the latter one seems to have a stronger influence on the fishery.

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1. Introduction

Fishery data is widely used to gather important information regarding the ecology of species [1–3]. Such data is also crucial to understand stock fluctuations and changes in the inherent ecosystem to help with the conservation and restoration of the resource as well as preventing a sudden collapse in the stock [2]. Especially for species exhibiting aggregation behaviour, fisheries data are a valuable source of information since species aggregations are commonly targeted by fishers and provide a predictable opportunity for large catches and an easy source of income [4,5]. While fishing on aggregations may produce short-term economic benefits to fishers, this activity also puts exploited fish stocks at risk of overfishing leading to a possible permanent loss of the resources [4,5].

The study of fishery data is particularly critical in systems that have a long history of exploitation, such as the Mediterranean Sea [6,7]. For instance, in the Balearic Islands (Spain), historical landing data was used to comprehend fluctuations in the *Coryphaena hippurus* fishery [8]. The Mediterranean Sea is a semi-enclosed basin characterised by coastal small-scale fisheries, representing 83% of the entire Mediterranean and Black Sea fleet in 2020 [9]. However, the emergence of more technologically sophisticated vessels (trawlers and purse seine) and the development of tourism after the Second World War, had detrimental impacts on traditional artisanal fisheries [10]. From the 1960s to the mid-1980s, total landings in the Mediterranean increased, potentially due to the modernization of fisheries (technological advances increased the effort and catchability efficiency) and growing nutrient load from coastal discharge [11], which consequently boosted the ecosystem's productivity. In contrast, a decline in nutrient flux from imposed regulations after the mid-1980s resulted in both low productivity and high rates of exploitation, causing the collapse of landings in subsequent years [11,12].

Ecological patterns can be difficult to comprehend when using fishery data, such as catch rates in small-scale fisheries [13], because they are incorporated within highly adaptive and complex marine social-ecological systems (SES) [14–16], posing a challenge in distinguishing between cause and effect [17].

For this reason, a fishery should not be regulated solely by ecological and biological components. Instead, it should aim to incorporate other potentially significant elements, such as social and economic factors that can equally affect the fishery in order to strive for optimal fisheries management [7,18], as considered in this study when attempting to assess the efficiency of a Marine Protected Area [19].

Price is a clear example of how an economic variable can affect fishers' behaviour and fishing strategy, according to seasonal price variations. For instance, it was found that after the adoption of a quota for *C. hippurus* fishery in 2013, to increase fishers' revenue, landings in Mallorca went from being recorded only in summer and autumn, to starting appearing all year long [8]. Additionally, tourism can have a tremendous impact on fishing activities, species abundance and even market price value, as the most targeted species and fish of ideal size are in greater demand from restaurants during the high season [10,20,21]. Likewise, fisheries management measures such as regulations have the power to influence fishing effort and catch rates [8,14], and thus should also be considered when addressing the study of the ecology of target species [22,23]. Established quotas and closed seasons can help with conservation of the fishing resource while also improving the income of fishers and retailers by increasing the demand in periods when this resource is not allowed to be fished [8,24].

Since the 1950s, tourism has been the primary contributor to economic growth in the Balearic Islands and it is still considered one the primary sources of income for what is known to be one of the main tourist destinations in Europe [20,25]. During the high season (May to October), the warmer seas attract millions of tourists to these islands, leading to changes within the fishing sector [20], for example, higher market values [21] as a function of increased demand for fish resources during these months [10,20].

Small-scale fisheries have been historically present on the island of Mallorca, with 241 out of 273 boats attributed to the small-scale fisheries sector in the last century [20] and accounting for 27% of the total income from Mallorca's landings during 2000–2014 [26]. These statistics demonstrate the considerable social weight this fishing sector adds to the fishery, since it is responsible for several hundred jobs [7].

One illustrative example of a complex socio-ecological system is supported by the small-scale fishery occurring in the Balearic Islands around *Seriola dumerili* fishery. This species, also commonly known as greater amberjack, has a circum-global distribution in subtropical and temperate pelagic waters [27] usually associated with rocky reefs or drop-offs areas [28]. In the Mediterranean, this apex-predator exists in great numbers [29] and forms spawning aggregations between late-spring and early-summer [30–32] when it reaches around 110 cm [31]. Even though it has been registered at depths of 360 m [28,33], during reproduction this species moves towards inshore warm waters, near the coast [30]. In contrast, juveniles are observed in offshore waters usually aggregated to *Sargassum* and floating objects [27,28].

Due to its aggregating behaviour, during both its early life stages and as an adult [34], *S. dumerili* is prone to being exploited by fishers who focus their effort on these key life-history events, such as *purse seiners*. This puts the species at a potential risk of being overexploited, as were their relatives in the NW Gulf of Mexico that were found to be overfished as a result of increased fishing effort and landings [27].

S. dumerili is a popular target species for recreational and commercial fishers worldwide, with distinct economic value [8,35]. In the Balearic Islands, this species represents one of the fifth most valued fish resources, which underlines its great cultural and socio-economic importance to the Balearic community (Figure 1A). Moreover, landings of this important resource suggest a seasonal pattern of vulnerability during its life cycle (Figure 1B) to different fishing gears strategically deployed in space and time to capture this species, a fact that illustrates the potential effect on the fishery caused by both ecological and social factors.

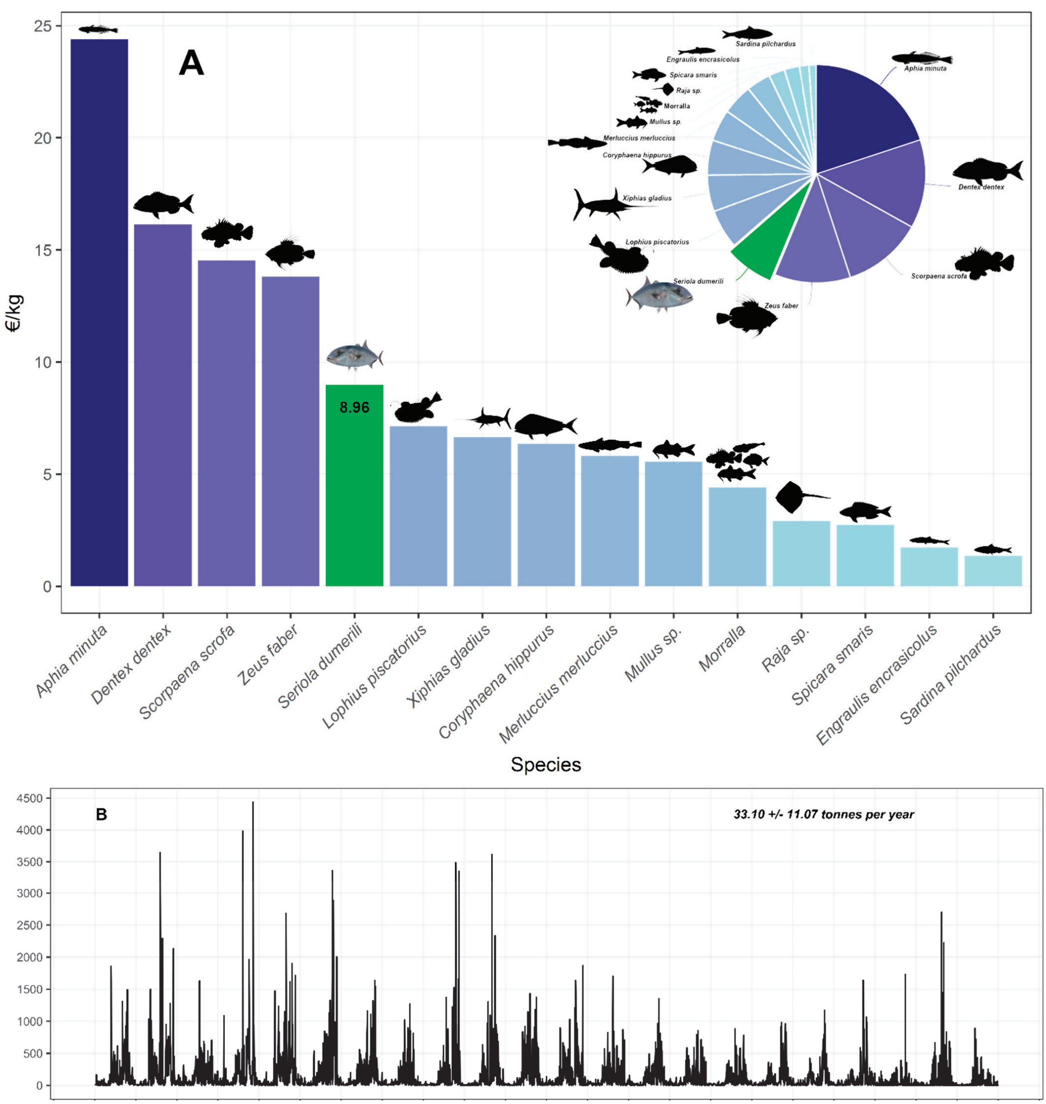

Figure 1. *Cont.*

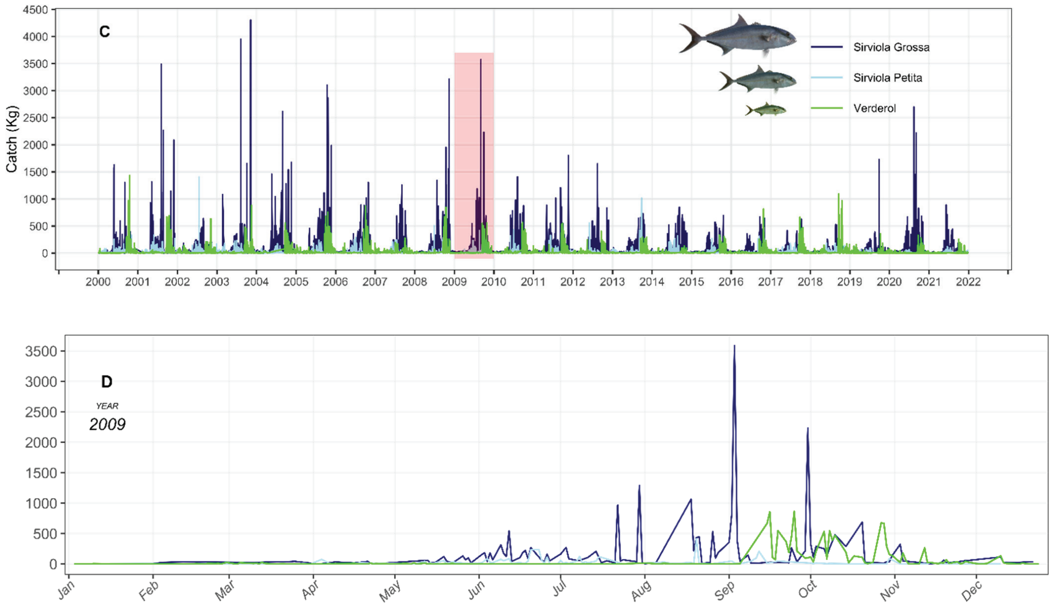

Figure 1. Socio-ecological data supporting the potential SES around *S. dumerili* fishery. (**A**) Economic importance of the main Balearic species as a function of price market value (€/Kg). Pie-chart represents the % of total price for each species as an illustration of economic importance. Graph from the Ministry of the Environment, Agriculture and Fisheries of the Government of the Balearic Islands, using the fishery statistics for 2015. Total daily catches of *S. dumerili* (**B**) and by size classes (**C**) *Sirviola Grossa* (navy blue), *Sirviola Petita* (light blue), *Verderol* (green) from the period 2000 to 2021. Monthly catch distribution from the year 2009 is represented in (**D**).

In an attempt to protect and ensure the sustainable management of this relevant fisheries resource, the Spanish Ministry of Agriculture, Food and Fisheries passed legislation in 2000 specifying a closed season for the fishery. This closed season applies to offshore waters of the archipelago with the aim of protecting *S. dumerili* juveniles, also known as *Verderol*, between the 1st of July and the 15th of September [36]. Later in 2011, an open fishing period was recognized for some species captured by purse seines, including *S. dumerili*, from 15th of July to 15th of November [37]. Finally in 2013, an exclusive open season extending from the 25th of August to the 31st of December was created in off-shore waters for the *Llampuga* fishery, a traditional small-scale fishery based on fishing aggregation devices (FAD) that targets *C. hippurus*, an epipelagic species fished alongside *S. dumerili* juveniles [8,38]. Such management strategies are focused on the protection of the recruitment phase and aggregations performed by this species. In Spain, the current major fisheries management body responsible for regulating the purse seine fishery, including the *S. dumerili* fishery in the Balearic Islands, is the national Ministry of the Environment, Rural and Marine Affairs [37].

Despite the important worldwide socio-ecological importance of this marine resource, little is known about the ecology of *S. dumerili* [35]. The majority of the studies focus on rearing and reproduction in captivity [39–41] with a view to incorporating it as an aquaculture species, given it rapid growth rate, commercial value, and good adaptation to captivity [42,43]. Therefore, due to the high commercial importance of this species, its vulnerable gregarious behaviour and the unknown effects of the imposed regulations, *S. dumerili* is at risk of overfishing with potential consequences for its conservation [44,45], hence the demand for urgent key information for an optimal evaluation and sustainable management.

In this sense, the main goal of the current study is to provide the first overview of the *S. dumerili* fishery from a socio-ecological framework, using the paradigmatic case study of this resource, which is intensively exploited by recreational and artisanal fleets in the Balearic Islands [45]. In this case-study, not only the ecological aspects such as spatio-temporal distribution (led by the spawning migrations [29] or aggregation events [33]) could moderate the exploitation of this highly-valued resource, but also the seasonal and by-size class prices and/or the fishery closed-season imposed, as potential social aspects, could interact to draw the landing patterns observed in the fishery.

This study strategically assesses the relationship with ecological and socio-economic factors of the complex SES around the *S. dumerili* fishery, through the means of historical and detailed landings data. Studying this relationship could significantly help the scientific community understand the currently scarce ecological information about *S. dumerili* within an exploited system. Such socio-ecological approaches might strongly contribute to fisheries management in the Mediterranean and to the understanding of fluctuations in catch and price rates while providing vital information for decision and policy makers to effectively and sustainably manage the fishery [45].

2. Materials and Methods

2.1. Study Area

This study was conducted in the Balearic Islands, a Spanish region, composed of four islands: Mallorca, Menorca, Ibiza and Formentera [46] (Figure 2). This archipelago is located in the NW Mediterranean between 38°35′–40°05′ N and 04°20′–01°15′ E [20] and it is recognized as an independent fisheries management unit [26,46]. Mallorca, with an area of 3620 km^2 and about 623 km of coastline [45], is the main island of the archipelago and is where most fish are landed and sold [20].

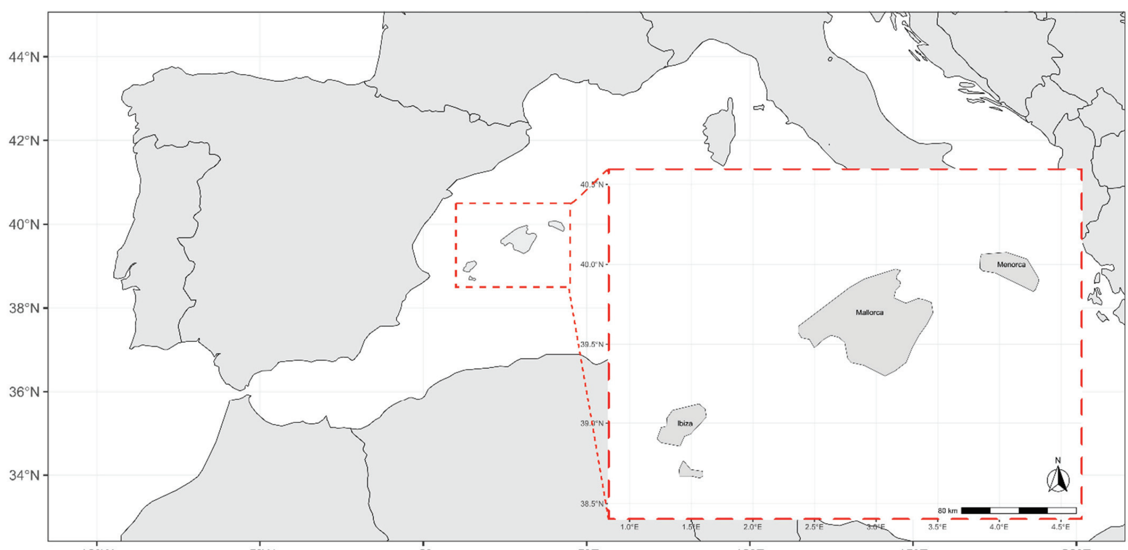

Figure 2. Overview map of the study area with main islands included: Mallorca, Menorca, Ibiza, and Formentera (below Ibiza).

2.2. Data Collection, Curation and Management

To gather information about the *S. dumerili* fishery, two different data sources were used during this study. Historical records of captures from the period 1950 until 2018 were obtained from *Sea Around Us* website (www.seaaroundus.org, accessed on 16 March 2022). This is an international project that began in 1999 in an attempt to estimate worldwide reported and unreported total fisheries catches [47]. Detailed landing data of Balearic fishing catches from the last two decades (2000–2021) were provided by the Palma Fishing Wharf in Opmallorcamar, Mallorca.

In order to provide an overview of the historical trend in *Seriola dumerili* catches (1950–2021), both datasets were combined. The period from 1950 to 1999 was covered by the historical timeseries from *Sea Around Us*, while for the last two decades (2000–2021) the Palma Wharf landing data was used. This last dataset contained extensive and detailed landing information for the last 21 years and included the daily weight of catches of *S. dumerili*, along with social and fishery information regarding the fishing vessel, gear and price per kg. Moreover, this data was available and subcategorized by the Palma Fishing Wharf in descending order of size (*Categories*), locally known as *Sirviola Grossa*, *Sirviola Petita* and *Verderol* with the latter including juveniles of the species that do not reach 30 centimetres [36]. *Sirviola Grossa*, may be considered to be all individuals larger than 100 cm since they represent adult specimens while *Sirviola Petita* sizes are those between the adults and juveniles. These categories are based on size, which can be related to the different life stages which are essential to provide information on how to properly disentangle the ecology of the species throughout the different phases of its life.

Two new categorical variables—*Regulation* and *Season*—were added to the data frame. The first was based on the law created in 2011, which established a closed fishing season (between 16th of November to 14th of July) for purse seines targeting *S. dumerili* [37], and the second one to account for potential seasonality patterns.

In order to describe the potential effect of *Regulation* on *S. dumerili* fishery, the yearly and monthly catch per unit effort (CPUE, kg/trip) was calculated by adding up the total catch of *S. dumerili* for every year and month, respectively, and dividing by the yearly and monthly number of fishing vessel trips. These estimations only accounted for purse seines given this fleet focuses on the exploitation of this resource as a target species (most important gear in terms of landings; see below in the results Section 3.2).

All data cleaning, exploration and analysis was performed using R version 4.2.0 [48]. Outliers were checked visually with the help of preliminary boxplots and kept for all variables except *Price*, to avoid removing values that could potentially be real catches and consequently create misleading results [49].

2.3. Statistical Analysis

The response variables, *Catch* and *Price*, were included in different generalized models as a function of various predictor variables, adjusting all statistical procedures according to the socio-ecological patterns found in *S. dumerli* fishery to be addressed (Table 1).

Table 1. Summary of models performed and their statistics. Respective response variables, potential transformations and explanatory variables used for each model.

	Response Variable	Explanatory Variables	Model	Transformations	r^2
Historical Trends	Catch (Kg)	Year	GAM (Catch ~ s(Year, k = 13)) [1]	Untransformed	0.689
Ecological Traits	Catch (Kg)	Categories Season	GLM (Catch ~ Categories * Season) [1]	Ordered Quantile Normalization (ORQ)	0.268
	Catch (Kg)	Gear	GLM (Catch ~ Gear)	Ordered Quantile Normalization (ORQ)	0.174
Socio–Economic Traits	Price (€/Kg)	Categories Month	GAM (Price ~ Categories + s(Month, k = 4) + s(Month, Categories, k = 7) [1]	Untransformed	0.315

[1] Interactions terms are indicated with * while smoothed terms are represented by *s*, depending on the model conducted (GLMs and GAMs, respectively).

Both Generalized Linear (GLMs) and Generalized Additive Models (GAMs) were performed during this study. GLMs [50] were used to assess linear relationships between response and explanatory variables (*glm* function from *lme4* package [51]), as this is a model that allows incorporation of non-normal response variables [50]. Whenever a non-linear relationship was observed in preliminary graphs, between explanatory and response variables, GAMs (*mgcv* package) were chosen, due to their higher flexibility when accounting for smoothness of the terms in the model [52]. For both models, when worth considering, interactions between predictors were included. When the data showed non-normality and homoscedasticity, transformations were performed using *bestNormalize* package [53], which selected the best normalizing transformation for the available data.

To provide an overview of the historical trends of *S. dumerili* catches, *Catch* (weight of *S. dumerili* by year) was modelled as a function of the *Year* using GAM (Table 1) as a data smoothing procedure to reduce the yearly variation. Meanwhile, ecological traits were assessed using two GLMs (Table 1). Transformed values of *Catch* (weight of *S. dumerili* for a given day and gear) were fitted as a function of the interaction of the categorical variables *Categories* and *Season* in order to disentangle a potential seasonal pattern of capture using the different life-stages of the species. Secondly, the same transformed response variable *Catch* was modelled as a function of *Gear*, in an attempt to relate the ecology of the species based on the fishing behaviour.

To disentangle a potential social effect of the price on the seasonal landings pattern of *S. dumerili* size classes, *Price* (€/Kg of *S. dumerili* daily sales) was fitted as a function of the categorical and numerical variables, *Categories* and *Months*, respectively, and the interaction of both using a GAM (Table 1).

In relation to the GAMs, the selection of the basis dimension parameter *k* was based on the diagnostic tests from *gam.check* function included in the *mgcv* library [54] and with the purpose of avoiding model overfitting (Table 1). A gamma value of 1.4 was also included in both GAMs for the same reason. The effective degrees of freedom (*edf*), when close to 1, was indicative of linearity between stressor and response variables [55].

For all models, the response variables were assumed to follow a Gaussian distribution with an identity-link function. Moreover, the goodness-of-fit was assessed as a function of the Akaike Information Criterion (AIC) [56], with significance of the variables and deviance explained. The distributions of the residuals were subsequently checked for normality with *qqplots* and confirmed among residuals, while the R *effects* library [57] was used to obtain the partial effects of the fixed factors. Predicted mean prices for all size categories were fitted with the function *predict_gam* of *tidymv* package and plotted using *ggplot2* library.

3. Results

3.1. Historical Trends of S. dumerili Catches

Record of catches of *Seriola dumerili* in the Balearic Islands region were analysed for a total period of 72 years (1950–2021) (Figure 3). After 1960, catches of *S. dumerili* considerably increased, reaching a peak of approximately 71 tonnes in 1969. This trend was followed by a continuous decline until the present day, with the lowest catch being recorded in 2015 (Figure 3).

Catch showed significant inter-annual variability among years ($p < 0.05$), with record values in the mid-1960s and following a decline after the 1980s. Catches increased again in the mid-2000s but immediately dropped straight after, until now (Figure 3). The residuals of the model revealed normality.

3.2. Ecological Factors

Landings of *S. dumerili* from the period 2000 to 2021 registered a mean of 33.10 ± 11.07 tonnes per year, with higher numbers occurring predominantly in the first decade (Figure 1B,C by size classes). Within a year, catches maintained low levels in the beginning until the start of the second semester, when landings of *S. dumerili* started increasing (Figure 1D), indicating a high seasonality pattern. *Sirviola Petita* (medium size) appeared throughout the year in very low and stable catches while *Sirviola Grossa* (bigger size) reached higher values in *Spring*, *Summer* and *Fall*, peaking in *Summer* months (Figure 4A,B; $p < 0.05$). Additionally, the landings of juveniles (*Verderol*) grew particularly during *Summer* and *Fall* (Figure 4C,D).

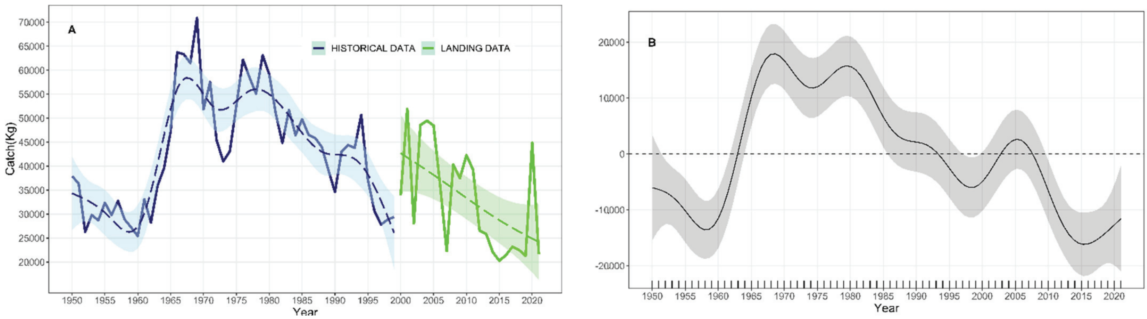

Figure 3. (**A**) Yearly distribution of *Catch* of *S. dumerili* for the period 1950–2021, obtained from both historical (blue) and landing (green) data. Dashed line is the smoothed trend estimated by the GAM method of the *geom_smooth* function. Blue and green shadows represent the confidence intervals. (**B**) GAM smooth splines of the response variable *Catch* as a function of the explanatory variable, *Year* and respective 95% confidence intervals.

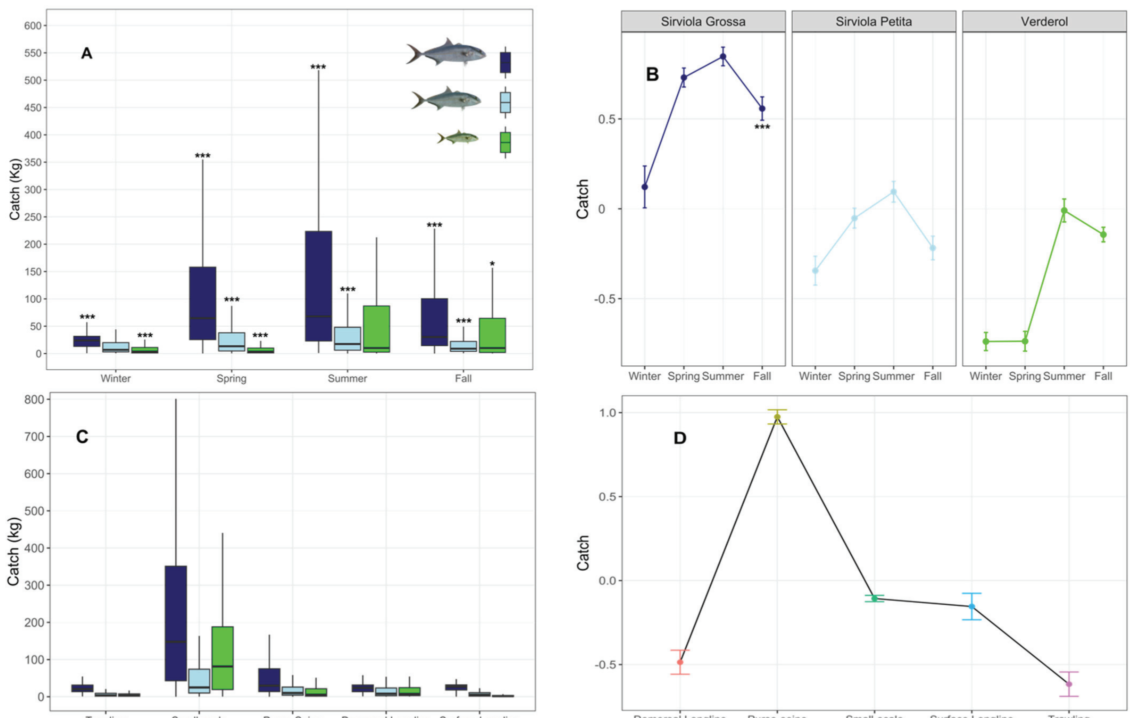

Figure 4. (**A**) Seasonal distribution of daily catches of the different size classes (asterisks above boxes indicates *p*-values * < 0.05 and *** < 0.001). (**B**) Partial effects of the interaction between *Categories* and *Season* on *S. dumerili* catches. (**C**) Proportion of *S. dumerili* landings (kg) for each *Gear* investigated with correspondent partial gear effect plot (**D**).

Catches from *Verderol* in *Summer* did not show significant effects ($p > 0.05$) (Figure 4A). The model resulted in a r^2 of 0.268 (Table 1) and residuals showed normality. Partial effects of the residuals revealed higher catches for *Sirviola Grossa* when compared to *Sirviola Petita* and *Verderol*, with the first two demonstrating the same seasonal pattern of landings (lower in *Winter*, peaking during summer followed by a decrease in *Fall*). Moreover, *Verderol* registered almost no catches during *Winter* and *Spring*, only appearing later in *Summer*, when it peaks (Figure 4A,B).

All gears studied revealed significant catches of *S. dumerili* ($p < 0.05$; Figure 4C,D). However, *Purse seine* is the gear that accounted for the majority of *S. dumerili* landing weight, with an average of approximately 16.1 tonnes per year ($p < 0.05$; Figure 4C,D) revealing a clear targeting pattern of *Seriola dumerili* by this fleet. In Majorca, *purse seiners* are multispecies, meaning that they target many different species throughout the year, such as tunas, squid, sparids, anchovies and sardines, while small-scale fisheries, responsible for being the second gear with the most catches, comprise a various number of passive gears that target *S. dumerili:* "solta" (February to April and September to December); "moruna" (April to September; similar to tuna traps deployed in shallow waters in search of spawner aggregations); "mussolera" (historically to catch demersal shark *Mustelus mustelus*: February to April) and "currican" (mainly *Verderol*: September to October). As a bycatch, *S.dumerili* can be caught with: "llampuguera" (mainly *Verderol*: August to November), "almadrabilla" (March and April) and trammel net (targeting spiny lobster *Palinurus elephas* and *Mullus* sp.) [58].

3.3. Socio-Economic Factors

Mean *Price* (€/Kg) of the different size categories of *Seriola dumerili* varied through seasons and across months ($p < 0.05$; Figure 5A,B). Although *Sirviola Grossa* maintained similar prices throughout the year, *Sirviola Petita* and *Verderol* disclosed seasonal patterns indicating an increase of economic value during *Summer* months (Figure 6B), reaching a maximum of 18.5 €/Kg and 19.1 €/Kg in August and July, respectively.

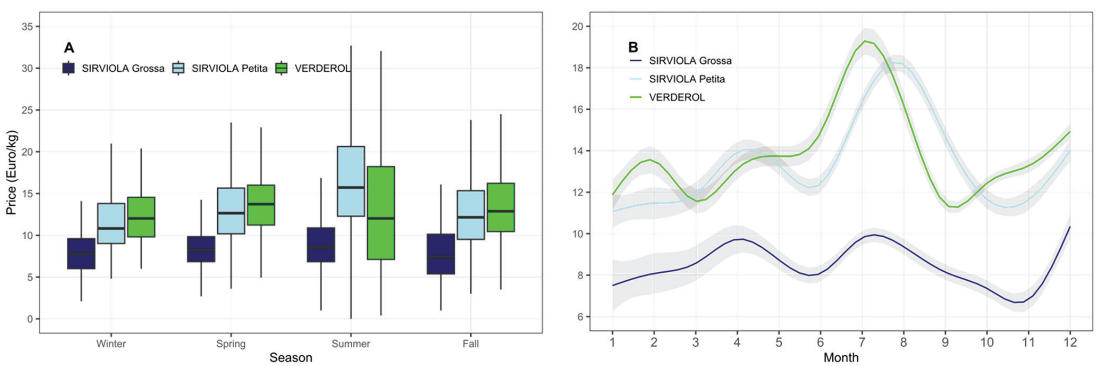

Figure 5. (**A**) Mean price (€/Kg) for all three size categories by *Season* (**B**) GAM predicted smooth splines of the response variable mean price of the different categories as a function of month ($p < 0.001$).

The GAM prediction of *Price* trend along the *Months* of the year confirmed the pattern first observed with the exploratory plots by demonstrating significant effects among *Categories* ($p < 0.05$) mean prices and through months. The predicted values of *Price* displayed, once again, a clear seasonal relationship between mean prices of *Verderol* and *Sirviola Petita* and were less prominent for *Sirviola Grossa*. These two categories of *S.dumerili* reached higher prices during optimal months of the year (July and August), after which they decrease back to lower prices (Figure 5B).

Similar CPUEs levels before and after the *Regulation* were shown for the last twenty years (Figure 6A). An increase CPUE trend along the months, peaking in August and followed by a consecutive decline, was detected (Figure 6B). Furthermore, such inter-month trends seemed to have remained fairly similar for the period affected (after 2011) and not affected (before 2011) by the *Regulation* (Figure 6B). For the months where fishing was not always allowed (January to May and December), CPUEs remained at relatively low numbers regardless of whether the regulation was enforced or not. For instance, in December there were no fishing trips occurring after 2011 while in April only one fishing trip was conducted for the same period (Figure 6B).

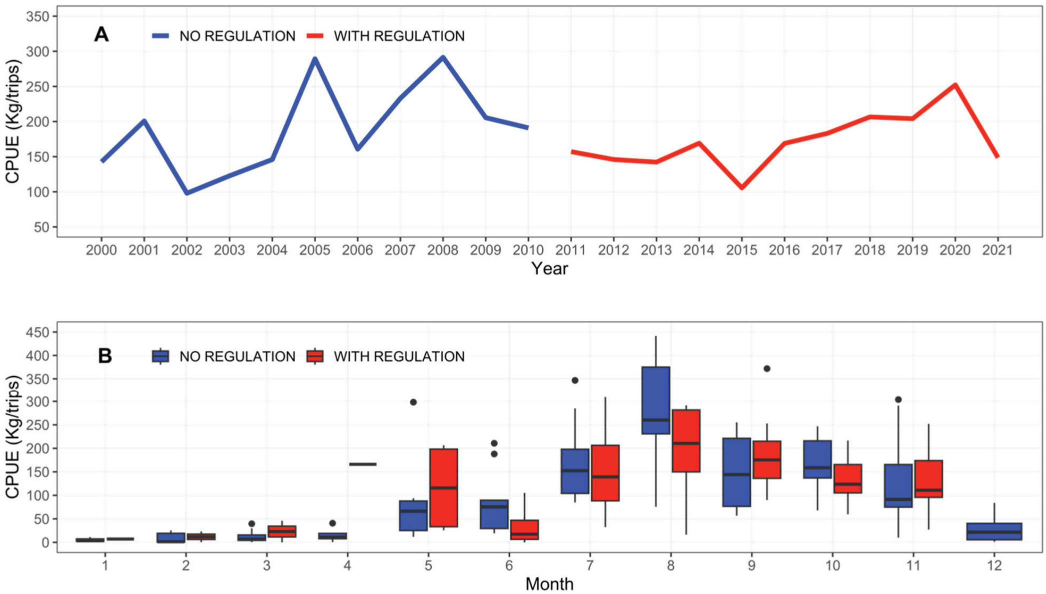

Figure 6. (**A**) CPUE distribution (2000–2021), for purse seine fleet that only target *S. dumerili*, as a function of the Regulation, (blue—before 2011; red—after 2011). (**B**) Monthly CPUE distribution, as a function of the Regulation (blue—before 2011; red—after 2011).

4. Discussion

This study represents the first attempt at investigating the *Seriola dumerili* fishery and its potential relationship with ecological and socio-economic factors. By analysing over 72 years of landing data in the Balearic Islands, evidence was found that the fishery was influenced by more than just biological components, with social and economic aspects holding regulating power as well.

Whilst reconstructed Balearic catch data between 1950 and 1999 [46] should be interpreted with caution, nonetheless, the trends observed seem to follow the common historical fishery. The big increase registered in the second half of the last century, near the 1960s, was a common pattern reported among Mediterranean fisheries [46]. The Balearic's increase of *S. dumerili* captures could be closely related to the expansion of consumption resulting from tourism and technological improvements that allowed fishing gear to become more efficient and consequently increase vessel's fishing power [10,59]. The declining trend from the 1980s can either be indicative of stock overexploitation, a decrease of fishing capacity owing to a decay in the number of fishing vessels [26,60,61] or a combination of both. Another possible explanation may be the decline of nutrient flux registered in the Mediterranean after the mid-1980s as reported by some studies [11,12]. This development in combination with high rates of exploitation might have led to the collapse of landings [6]. The declining trend has been prominent until the present day, although there was a slight increase in the mid-2000s but never comparing to levels documented in the past century.

Within this decreased scenario of landings, our results strongly suggest a seasonality pattern in landings of different life-stages, probably related to the species ecology. In fact, *S. dumerili* spawners (in most part *Sirviola Grossa*) undergo spawning aggregations between late-spring and early-summer in the Mediterranean [30,31], periods in which this species becomes more susceptible to being caught [44,62–64]. The non-significance of catches of juveniles in *Summer* can either be related to the fact that there are similar catches of *Verderol* during *Fall* or be in comparison with other size *Categories* such as *Sirviola Petita*, which has identical numbers of catches in *Summer*. Nevertheless, the increase of juveniles catches during this period is probably a consequence of the species recruitment [34,65], as they can become available for the fishery faster due to their rapid growth throughout early life stages (1.45 mm/day) and for late season cohort [27]. Meanwhile, the medium size fish (*Sirviola Petita*) was found at relatively low levels between seasons, with slightly higher catches occurring in *Summer*, just as for *Sirviola Grossa*, and possibly related to the behaviour and ecology of the species. A point to be highlighted is the lower record of landings during *Winter* months (January–March) for all three size categories. Social and economic factors could be influencing this tendency [8], however based on the results obtained in the present study, these socio-economical aspects can be discarded, considering the spatio-temporal behaviour and ecology of this species. Other fisheries, such as the transparent goby (*Aphia minuta)* fishery, register high catches during these cold months (December to April), while the dolphinfish (*C. hippurus)* fishery sustains low catches for the first two trimesters of the year (January to August), demonstrating a clear seasonality pattern in the species caught by small-scale fisheries. Besides, this colder period falls outside the temporal spawning-window [32] and therefore, a possible disaggregate schooling behaviour and migrations towards deeper waters [30] could explain the decreased availability of the resource for the fishery.

This explanation is in agreement with the fishing behaviour of the purse seine fleet which accounted for 56% of total catches. Just as for other central Mediterranean cities, purse seines stood out as the main contributor for most *S. dumerili* landings [30]. This was already expected since purse seines' main strategy is to use encircling nets to target species that are aggregated in a specific place [66]. Moreover, as mentioned above, *S. dumerili* is a reef-associated species [34] with a clear aggregating behaviour during spawning events [30] and in earlier stages, as juveniles are also known to aggregate in offshore areas starting in July [33,34,65]. Such spatio-temporal schooling behaviour during different life stages of *S. dumerili* ultimately makes this species more vulnerable and likely to be exploited by the purse seine fishery, who seem to focus their exploitation efforts on the time and place the species is available and economically profitable [61].

Regarding socio-economic factors, price and its link with tourism showed an effect on landings of the fishery, while fishing regulation seems to have had no effect. The rise for these two size categories' (*Sirviola Petita* and *Verderol*) prices strongly matches the touristic high season in the Balearic Islands [20,21]. With the increasing arrival of tourists in the archipelago, the demand grows in restaurants, hotels and local fish markets, with higher preference for small to medium average size fish, in detriment of larger fish. This pattern explains the relatively constant low price trend for *Sirviola Grossa* across the year [10,20,21]. After peaking, *Sirviola Petita* and *Verderol* mean prices suffer a steep decline, probably due to the end of the intense high season, or as a consequence of market value change, coinciding with the period of time *Verderol* becomes most vulnerable to being harvested and therefore a higher availability of this resource results in lowering prices (a clear marketing example of the supply and demand law [67]). Another social aspect that clearly influences fisheries in term of landings and spatio-temporal distributions of effort are regulations [14]. However, similar CPUEs between periods of regulation and no regulation may prove that even though socio-economic factors have the power to affect this fishery [61], it seems that the ecological aspects hold more power when it comes to influencing *Seriola dumerili* fishery.

A detailed understanding of the ecology of *S. dumerili* is fundamental to its conservation and resource management considering the economic importance of this species both in the Balearic Islands and worldwide [2]. Little is still known about *S. dumerili* ecology, as the main study focus has been on captivity rearing and reproduction [68–71]. This study provides key socio-ecological insights on *S. dumerili* fishery; however, it is vital to conduct future research to improve the successful management of this resource in the Balearic Islands and Mediterranean Sea.

Analysing historical data of catch per unit of effort would shine light on *S. dumerili* fisheries' history and help assess the current status of the stock, while telemetry experiments (e.g., acoustic and satellite transmitters [72]) could provide important information about the species distribution (previously conducted in other places with related species *Seriola rivoliana* [1]) and confirm the hypothesis of offshore spawning migrations potentially being behind the lower catches of *S. dumerili* during winter months. Additionally, as this fishery is inserted in a system with complex human–environment relationships and profound uncertainties [16], it is highly recommended to deepen the knowledge on other potential ecological and socio-economic indicators, such as the population structure and growth, reproductive traits, employment and income.

Finally, and since the current legislation did not appear to have an effect on CPUE of *S. dumerili*, an extension of the closed season (from May to August) would fully cover the spawning period of this species. Despite being suggested a short spawning period (May–June) triggered mainly by higher temperature cues [32], the gradual increase of such a variable under the tropicalization of the Mediterranean Sea [73] could expand the temporal window of the spawning. Therefore, this conservative measure (extended closed season from May to August) might ensure the protection of this important life-history trait and vulnerable event for *S. dumerili* [34] with potential consequences on its population dynamics. It is important to ensure that future regulations made by decision and policy makers are socially and ecologically fair [3] and supported by the best available information to effectively and sustainably manage the *S. dumerili* fishery [45].

5. Conclusions

The present study represents the first research conducted providing an overview of the *S. dumerili* fishery from a socio-ecological framework. The findings of this study contribute to a better understanding of the ecological, economic and social pillars of *S. dumerili* fishery. After analysing historical and detailed landing data, results revealed an important seasonal fishery occurring in the NW Mediterranean, mainly influenced by the species ecology with considerable weight given to socio-economic factors. As such, this study provides essential information for proper sustainable management of one of the most important marine resources not only in the Balearic Islands but across the world.

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References

1. Fontes, J.; Schmiing, M.; Afonso, P. Permanent aggregations of a pelagic predator at shallow seamounts. *Mar. Biol.* **2014**, *161*, 1349–1360. [CrossRef]
2. Fromentin, J. Lessons from the past: Investigating historical data from bluefin tuna fisheries. *Fish Fish.* **2009**, *10*, 197–216. [CrossRef]
3. Nataniel, A.; Lopes, P.F.M.; Lopez, J.; Soto, M. Socio-ecological and economic aspects of tropical tuna fisheries in the Mozambique Channel. *Fish. Manag. Ecol.* **2022**, *29*, 115–130. [CrossRef]
4. de Mitcheson, Y.S.; Colin, P.L. *Reef Fish Spawning Aggregations: Biology, Research and Management*; Springer: Berlin/Heidelberg, Germany, 2012. [CrossRef]
5. Erisman, B.; Heyman, W.; Kobara, S.; Ezer, T.; Pittman, S.; Aburto-Oropeza, O.; Nemeth, R.S. Fish spawning aggregations: Where well-placed management actions can yield big benefits for fisheries and conservation. *Fish Fish.* **2017**, *18*, 128–144. [CrossRef]
6. Colloca, F.; Scarcella, G.; Libralato, S. Recent trends and impacts of fisheries exploitation on Mediterranean stocks and ecosystems. *Front. Mar. Sci.* **2017**, *4*, 244. [CrossRef]
7. Farrugio, H.; Oliver, P.; Biagi, F. An overview of the history, knowledge, recent and future trends in the Mediterranean fisheries. *Sci. Mar.* **1993**, *57*, 105–119.
8. Camiñas, J.A.; Grau, A.M.; Hernández, P. GCP/INT/028/SPA—GCP/INT/006/EC. CopeMed II Occasional Paper Nº 21. 2016. Available online: https://www.fao.org/fishery/en/publications/301206 (accessed on 3 March 2022).
9. FAO. *The State of Mediterranean and Black Sea Fisheries*; FAO: Rome, Italy, 2020.
10. Gómez, S.; Lloret, J.; Demestre, M.; Riera, V. The Decline of the Artisanal Fisheries in Mediterranean Coastal Areas: The Case of Cap de Creus (Cape Creus) The Decline of the Artisanal Fisheries in Mediterranean Coastal Areas: The Case of Cap de Creus (Cape Creus). *Coast. Manag.* **2006**, *34*, 217–232. [CrossRef]
11. Fortibuoni, T.; Giovanardi, O.; Pranovi, F.; Raicevich, S.; Solidoro, C.; Libralato, S. Analysis of long-term changes in a mediterranean marine ecosystem based on fishery landings. *Front. Mar. Sci.* **2017**, *4*, 33. [CrossRef]
12. Fiorentini, L.; Caddy, J.F.; de Leiva, J.I. Long and short-term trends of Mediterranean fishery resources. *Stud. Rev. Gen. Fish Counc. Mediterr.* **1997**, *69*, 72.
13. Hidalgo, M.; Rouyer, T.; Molinero, J.C.; Massutí, E.; Moranta, J.; Guijarro, B.; Stenseth, N.C. Synergistic effects of fishing-induced demographic changes and climate variation on fish population dynamics. *Mar. Ecol. Prog. Ser.* **2011**, *426*, 1–12. [CrossRef]
14. Ammar, Y.; Voss, R.; Niiranen, S.; Blenckner, T. Quantifying socio-economic novelty in fisheries social-ecological systems. *Fish Fish.* **2022**, *23*, 445–461. [CrossRef]
15. Naranjo-Madrigal, H.; van Putten, I.; Norman-López, A. Understanding socio-ecological drivers of spatial allocation choice in a multi-species artisanal fishery: A Bayesian network modeling approach. *Mar. Policy* **2015**, *62*, 102–115. [CrossRef]
16. Schlueter, M.; Mcallister, R.R.; Arlinghaus, R.; Bunnefeld, N.; Eisenack, K.; Hoelker, F.; Milner-Gulland, E.J.; Müller, B.; Nicholson, E.; Quaas, M.; et al. New horizons for managing the environment: A review of coupled social-ecological systems modeling. *Nat. Resour. Model.* **2012**, *25*, 219–272. [CrossRef]
17. Cochrane, K.L. Complexity in fisheries and limitations in the increasing complexity of fisheries management. *ICES J. Mar. Sci.* **1999**, *56*, 917–926. [CrossRef]
18. Woods, P.J.; Macdonald, J.I.; Bárðarson, H.; Bonanomi, S.; Boonstra, W.J.; Cornell, G.; Cripps, G.; Danielsen, R.; Färber, L.; Ferreira, A.S.A.; et al. A review of adaptation options in fisheries management to support resilience and transition under socio-ecological change. *ICES J. Mar. Sci.* **2022**, *79*, 463–479. [CrossRef]
19. Li, Y.; Sun, M.; Ren, Y.; Chen, Y. Impact of pre-closure fishing effort on marine protected area performance in social-ecological dimensions: Implications for developing marine conservation plans. *Sci. Total Environ.* **2020**, *729*, 138936. [CrossRef]
20. Iglesias, M.; Massutí, E.; Reñones, O.; Morales-Nin, B. Three small-scale fisheries based on the island of Majorca (NW Mediterranean). *Monogr. Boll. Soc. Hist. Nat. Balear.* **1994**, *37*, 35–38. Available online: http://hdl.handle.net/10508/6794 (accessed on 3 March 2022).
21. Merino, G.; Morales-nin, B.; Maynou, F.; Grau, A.M. Assessment and bioeconomic analysis of the Majorca (NW Mediterranean) trammel net fishery. *Aquat. Living Resour.* **2008**, *21*, 99–107. [CrossRef]
22. Arlinghaus, R.; Cooke, S.J.; Potts, W. Towards resilient recreational fisheries on a global scale through improved understanding of fish and fisher behaviour. *Fish. Manag. Ecol.* **2013**, *20*, 91–98. [CrossRef]
23. Hunt, L.M.; Sutton, S.G.; Arlinghaus, R. Illustrating the critical role of human dimensions research for understanding and managing recreational fisheries within a social-ecological system framework. *Fish. Manag. Ecol.* **2013**, *20*, 111–124. [CrossRef]
24. Moltó, V.; Hernández, P.; Sinopoli, M.; Besbes-Benseddik, A.; Besbes, R.; Mariani, A.; Gambin, M.; Alemany, F.; Morales-Nin, B.; Grau, A.M.; et al. A Global Review on the Biology of the Dolphinfish (*Coryphaena hippurus*) and Its Fishery in the Mediterranean Sea: Advances in the Last Two Decades. *Rev. Fish Sci. Aquac.* **2020**, *28*, 376–420. [CrossRef]
25. Balearic Islands Tourism Board. Balearic Islands Regional Context Survey. 2017. Available online: https://www.interregeurope.eu/fileadmin/user_upload/tx_tevprojects/library/file_1508251726.pdf (accessed on 27 March 2022).
26. Quetglas, A.; Merino, G.; González, J.; Ordines, F.; Garau, A.; Grau Jofre, A.M.; Guijarro, B.; Oliver, P.; Massutí, E. Regional Implementation Plan for Demersal Fisheries from the Balearic Islands (Western Mediterranean). 2016. Available online: https://doi.org/10.13140/RG.2.2.22245.81129 (accessed on 22 March 2022).

27. Wells, R.J.D.; Rooker, J.R. Distribution, age, and growth of young-of-the-year greater amberjack (*Seriola dumerili*) associated with pelagic Sargassum. *Fish Bull.* **2004**, *102*, 545–554.
28. FAO. *Seriola dumerili* (Risso, 1810). Available online: https://www.fao.org/3/y4162e/y4162e09.pdf (accessed on 5 March 2023).
29. Kozul, V.; Skaramuca, B.; Kraljevic, M.; Dulcic, J.; Glamuzina, B. Age, growth and mortality of the Mediterranean amberjack *Seriola dumerili (Risso 1810)* from the South-Eastern Adriatic Sea. *J. Appl. Ichthyol.* **2001**, *17*, 134–141. [CrossRef]
30. Andaloro, F.; Pipitone, C. Food and feeding habits of the amberjack, *Seriola dumerili* in the Central Mediterranean Sea during the spawning season. *Cah. Biol. Mar.* **1997**, *38*, 91–96.
31. FAO. *Seriola dumerili*. Cultured Aquatic Species Information Programme (FAO, Rome: Italy) Updated 2016-01-18. Available online: https://www.fao.org/fishery/en/culturedspecies/seriola_dumerili?lang=en (accessed on 20 April 2022).
32. Grau, A.M. Aspectos Histológicos, Ciclo Reproductor y Principales Procesos Patológicos de Seriola dumerili, Risso 1810 (Carangidae). 1992. Available online: https://dialnet.unirioja.es/servlet/tesis?codigo=267842 (accessed on 5 March 2022).
33. Froese, R.; Pauly, D. FishBase. World Wide Web Electronic Publication. 2022. Available online: www.fishbase.org (accessed on 29 May 2022).
34. Sinopoli, M.; D'Anna, G.; Badalamenti, F.; Andaloro, F. FADs influence on settlement and dispersal of the young-of-the-year greater amberjack (*Seriola dumerili*). *Mar. Biol.* **2007**, *150*, 985–991. [CrossRef]
35. Thompson, B.A.; Beasley, M.; Wilson, C.A. Age distribution and growth of greater amberjack, *Seriola dumerili*, from the north-central Gulf of Mexico. *Fish Bull.* **1999**, *97*, 362–371.
36. BOE-A-2000-7719. Available online: https://www.boe.es/buscar/doc.php?id=BOE-A-2000-7719 (accessed on 4 April 2022).
37. BOE-A-2011-15094. Available online: https://www.boe.es/diario_boe/txt.php?id=BOE-A-2011-15094 (accessed on 4 April 2022).
38. BOE-A-2013-9767. Available online: https://www.boe.es/eli/es/o/2013/09/10/aaa1688 (accessed on 4 April 2022).
39. Fernández-Montero, A.; Caballero, M.J.; Torrecillas, S.; Tuset, V.M.; Lombarte, A.; Ginés, R.R.; Izquierdo, M.; Robaina, L.; Montero, D. Effect of temperature on growth performance of greater amberjack (*Seriola dumerili*, Risso 1810) Juveniles. *Aquac. Res.* **2018**, *49*, 908–918. [CrossRef]
40. Hamasaki, K.; Tsuruoka, K.; Teruya, K.; Hashimoto, H.; Hamada, K.; Hotta, T.; Mushiake, K. Feeding habits of hatchery-reared larvae of greater amberjack *Seriola dumerili*. *Aquaculture* **2009**, *288*, 216–225. [CrossRef]
41. Stewart, D.S. Hatchery production of yellowtail kingfish (*Seriola lalandi*). In *Advances in Aquaculture Hatchery Technology*; Woodhead Publishing: Sawston, UK, 2013; pp. 542–553. [CrossRef]
42. Nakada, M.K. Yellowtail Culture Development and Solutions for the Future. *Rev. Fish Sci.* **2002**, *10*, 559–575. [CrossRef]
43. Honebrink, R.R. A review of the biology of the family *Carangidae*, with emphasis on species found in Hawaiian waters. *Div. Aquat. Resour. Tech. Rep.* **2000**, *20*, 159–169.
44. De Mitcheson, Y.S. Mainstreaming Fish Spawning Aggregations into Fishery Management Calls for a Precautionary Approach. *Bioscience* **2016**, *66*, 295–306. [CrossRef]
45. Morales-Nin, B.; Moranta, J.; García, C.; Tugores, M.P.; Grau, A.M.; Riera, F.; Cerda, M. The recreational fishery off Majorca Island (western Mediterranean): Some implications for coastal resource management. *ICES J. Mar. Sci.* **2005**, *62*, 727–739. [CrossRef]
46. Carreras, M.; Coll, M.; Quetglas, A.; Goñi, R.; Pastor, X.; Cornax, M.J.; Iglesia, M.; Massuti, E.; Oliver, P.; Aguilar, R.; et al. Estimates of Total Fisheries Removal for the Balearic Islands (1950–2010). 2015. Available online: http://www.seaaroundus.org/doc/publications/wp/2015/Carreras-et-al-Balearic-Islands.pdf (accessed on 7 April 2022).
47. Zeller, D.; Palomares, M.D.; Tavakolie, A.; Ang, M.; Belhabib, D.; Cheung, W.W.; Lam, V.W.; Sy, E.; Tsui, G.; Zylich, K.; et al. Still catching attention: *Sea Around Us* reconstructed global catch data, their spatial expression and public accessibility. *Mar. Policy* **2016**, *70*, 145–152. [CrossRef]
48. R Core Team. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2022; Available online: https://www.r-project.org/ (accessed on 16 April 2022).
49. Zuur, A.F.; Ieno, E.N.; Elphick, C.S. A protocol for data exploration to avoid common statistical problems. *Methods Ecol. Evol.* **2010**, *1*, 3–14. [CrossRef]
50. McCullagh, P.; Cheek, P.J.; Nelder, J.A. Generalized Linear Models. *Appl Stat.* **1990**, *39*, 385. [CrossRef]
51. Bates, D.; Machler, M.; Bolker, B.; Walker, S. Fitting Linear Mixed-Effects Models Using lme4. *J. Stat. Softw.* **2015**, *67*, 1–48. [CrossRef]
52. Wood, S.N. Generalized Additive models. *Stat. Sci.* **1986**, *1*, 297–310. [CrossRef]
53. Peterson, R.A. Finding Optimal Normalizing Transformations via best Normalize. *R J.* **2021**, *13*, 310–329. [CrossRef]
54. Wood, S.N. *Generalized Additive Models: An Introduction with R*, 2nd ed.; Chapman and Hall/CRC: Boca Raton, FL, USA, 2017.
55. Hunsicker, M.E.; Kappel, C.V.; Selkoe, K.A.; Halpern, B.S.; Scarborough, C.; Mease, L.; Amrhein, A. Characterizing driver-response relationships in marine pelagic ecosystems for improved ocean management. *Ecol. Appl.* **2016**, *26*, 651–663. [CrossRef]
56. Akaike, H. A New Look at the Statistical Model Identification. *IEEE Trans. Automat. Contr.* **1974**, *19*, 716–723. [CrossRef]
57. Fox, J.; Weisberg, S. *An R Companion to Applied Regression*, 3rd ed.; Sage: Newbury Park, CA, USA, 2019; Available online: https://socialsciences.mcmaster.ca/jfox/Books/Companion/index.html (accessed on 16 April 2022).
58. Mallol, S.; Goñi, R. *Seguimiento de Las Pesquerías (Punto Cero) En El Ámbito de La Futura Reserva Marina de Cala Rajada (Mallorca)*; IEO-COB y SGPM/PESCALA I/04-1: Pescara, Italy, 2004; 129p.
59. Murray, I. The fishing footprint of a tourism-based economy: Displacing seafood consumption from local to distant waters in the Balearic Islands. *J. Polit. Ecol.* **2015**, *22*, 211–238. [CrossRef]

60. Maynou, F. Evolution of fishing capacity in a Mediterranean fishery in the first two decades of the 21st c. *Ocean Coast. Manag.* **2020**, *192*, 105–190. [CrossRef]
61. Maynou, F.; Morales-Nin, B.; Cabanellas-Reboredo, M.; Palmer, M.; García, E.; Grau, A.M. Small-scale fishery in the Balearic Islands (W Mediterranean): A socio-economic approach. *Fish Res.* **2013**, *139*, 11–17. [CrossRef]
62. Robinson, J.; Cinner, J.E.; Graham, N.A.J. The influence of fisher knowledge on the susceptibility of reef fish aggregations to fishing. *PLoS ONE* **2014**, *9*, e91296. [CrossRef]
63. Sadovy, Y.; Domeier, M. Are aggregation-fisheries sustainable? Reef fish fisheries as a case study. *Coral Reefs* **2005**, *24*, 254–262. [CrossRef]
64. van Overzee, M.J.H.; Rijnsdorp, A.D. Effects of fishing during the spawning period: Implications for sustainable management. *Rev. Fish Biol. Fish.* **2015**, *25*, 65–83. [CrossRef]
65. Deudero, S.; Merella, P.; Morales-Nin, B.; Massutí, E.; Alemany, F. Fish communities associated with FADs. *Sci. Mar.* **1999**, *63*, 199–207. [CrossRef]
66. NOAA. Fishing Gear: Purse Seines. 2019. Available online: https://www.fisheries.noaa.gov/national/bycatch/fishing-gear-purse-seines. (accessed on 9 May 2022).
67. Gale, D. The Law Of Supply And Demand. *Math. Scand.* **1955**, *3*, 155–169. [CrossRef]
68. Jerez, S.; Samper, M.; Santamaría, F.J.; Villamandos, J.E.; Cejas, J.R.; Felipe, B.C. Natural spawning of greater amberjack (*Seriola dumerili*) kept in captivity in the Canary Islands. *Aquaculture* **2006**, *252*, 199–207. [CrossRef]
69. Corriero, A.; Wylie, M.J.; Nyuji, M.; Zupa, R.; Mylonas, C.C. Reproduction of greater amberjack (*Seriola dumerili*) and other members of the family *Carangidae*. *Rev. Aquac.* **2021**, *13*, 1781–1815. [CrossRef]
70. Papandroulakis, N.; Mylonas, C.C.; Maingot, E.; Divanach, P. First results of greater amberjack (*Seriola dumerili*) larval rearing in mesocosm. *Aquaculture* **2005**, *250*, 155–161. [CrossRef]
71. Jerez, S.; Cejas, J.R.; Martin, M.V.; Santamaría, F.J.; Felipe, B.C. Growth Performance of *Seriola dumerili* Juveniles Fed on Different Feeding Frequencies. *Cent Ocean Canar.* **2014**, *2*, 483–484.
72. Hussey, N.E.; Kessel, S.T.; Aarestrup, K.; Cooke, S.J.; Cowley, P.D.; Fisk, A.T.; Harcourt, R.G.; Holland, K.N.; Iverson, S.J.; Kocik, J.F.; et al. Aquatic animal telemetry: A panoramic window into the underwater world. *Science* **2015**, *348*, 1255642. [CrossRef]
73. Bianchi, C.N.; Morri, C. Global sea warming and "tropicalization" of the Mediterranean Sea: Biogeographic and ecological aspects. *Biogeogr. J. Integr. Biogeogr.* **2003**, *24*. [CrossRef]

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Comparing the Size at Onset of Sexual Maturity of Edible Crab (*Cancer pagurus*, Cancridae) in Berwickshire and Northumberland

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Abstract: The literature suggests regional variations in the size at which sexual maturity is reached for commercially important edible crab (*Cancer pagurus*), worth GBP 74.3 million annually, which could have implications for regional fisheries management. Berwickshire and Northumberland are geographically divided by the Scotland and England border and remain within the Berwickshire and North Northumberland Coast SAC (Special Area of Conservation). Each are managed by differing fisheries authorities and Minimum Conservation Reference Sizes (MCRS). Morphometric measurements were recorded for each *C. pagurus* individual to categorise morphometric maturity using segmented regression, with gonadal maturity categorised using visual gonad characteristics and general linear model regressions to compare onset in sexual maturity. Results showed regional variations for gonadal maturity with males reaching sexual onset at a carapace width size of 108.5 mm in Berwickshire and 109.9 mm in Northumberland; females at a size of 126.8 mm in Berwickshire and 120.8 mm in Northumberland. This was also true for morphometric maturity based on chelae height, that males (141.1 mm) and females (134.7 mm) from Berwickshire were morphometrically mature at greater sizes than males (130.1 mm) and females (120.8 mm) from Northumberland. This study shows that the respective MCRS in both regions are appropriate for the *C. pagurus* populations, but implications for fisheries management could be present.

Keywords: *Cancer pagurus*; conservation; fisheries management; MCRS; minimum landing size

Key Contribution: The size at onset of sexual maturity for *Cancer pagurus* differs between Berwickshire and Northumberland, UK. When the regions are compared, males mature at smaller sizes in Berwickshire (108.5 mm) and females mature at smaller sizes in Northumberland (120.8 mm). These sizes are still appropriate for the current regional minimum conservation reference sizes, but implications for fisheries management could be present.

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1. Introduction

The edible crab (*Cancer pagurus*, Cancridae) fishery is the most valuable crab fishery in the UK and an important economic product for inshore commercial fishermen. Annual landings were in excess of 31,632 tonnes with a landing value of just under GBP 74.3 million in 2019 [1,2]. The fishery is subject to EU regulations that impose statutory Minimum Conservation Reference Sizes (MCRS) which prohibit the landing and selling of undersized individuals. The MCRS for *C. pagurus* differs between regions and in Scotland the MCRS is 150 mm (carapace width) (except Shetland Isle where it is 140 mm) [3] while Northumberland (specifically the Northumberland Inshore Fisheries and Conservation Authority), England, MCRS is 130 mm [4]. The primary aim of the MCRS at a management level is to enable individuals to reproduce at least once before harvesting [5].

During 2020, there were 62 active creel fishing vessels (<10 m in length) operating from Eyemouth harbour (55.873726, −2.086900) [6], 75 active vessels in Northumberland, and 11 active vessels (based in Scotland) operating on both sides of the border [2,6]. Landings and effort data are available for 2020; however, COVID-19 restrictions that were in place for 2020 likely affected the fishing effort and landings for this period and therefore are unlikely to adequately represent the region as a whole. In 2020, the *C. pagurus* fishery for Berwickshire was valued at GBP 567,000 and GBP 2.13 million within the Northumberland region, highlighting the economic importance of this fishery to these regions [2,6]. *C. pagurus* landings data for the Berwickshire region and Northumberland region can be seen for 2017–2019 in Table 1.

Table 1. Landings and value per annum of *C. pagurus* from 2017 to 2019 for both study regions.

Region	Landings and Value Per Annum (Tonne)/(£'000)		
	2017	**2018**	**2019**
Berwickshire	530/820	498/1014	463/1081
Northumberland	1057/1375	971/1598	949/2140

Note: Berwickshire values are taken from the Scottish sea fisheries statistics reports retrieved from https://www.gov.scot/publications/scottish-sea-fisheries-statistics-2018/documents/ (accessed on 5 May 2022). These values are a collation of landings in the main regional port of Eyemouth. Northumberland values were provided by NIFCA datasets.

Increased reports from local fishermen (pers comm), who fish both sides of the Scottish/English border, have suggested that the discrepancy in MCRS between regions, and the close proximity of local fishing fleets to the border, has resulted in cross-border fishing and landings. If this is the case, current MCRS measures between regions are unlikely to be effective as the removal of individuals deemed undersized in one region are landed in the other, further limiting the benefits of maintaining these individuals to provide brood stock. Regional approaches to management are the current norm in managing the edible crab stocks in Berwickshire and Northumberland; however, with reports of cross-border landings, a harmonised MCRS for Berwickshire and Northumberland could prevent such activity. Therefore, understanding the size at which an individual becomes mature in both regions will be beneficial to the fishery and advancement of the fishery management.

C. pagurus is a decapod species that has an international distribution spanning from Norway to Morocco, located at depths of up to 100 m [7]. Typically, this species resides in habitats consisting of coarse sediment and rock [7]. The literature suggests that the size at onset of sexual maturity (SOM) of localised populations of *C. pagurus* differs spatially, depending on different environmental factors such as habitat type, temperature, and depth [8,9]. This is also reflected by the variance in MCRSs around the UK coast (Table 2). Maturity can be categorised into four criteria—gonadal, morphometric, behavioural and functional [8]. Each focusses on different stages of the decapods progression through maturity—gonadal is the presence of developed testes for males and ovaries for females; morphometric the changes in growth of chelipeds in males and the abdomen and pleopods for females; behavioural includes the presence of sperm plugs in the females' oviducts as an act of copulation; functional is a combination of the other three indicated by presence of offspring [8]. The hepatopancreas is an essential organ that provides the individual the energy and nutrients required for growth and progressing through reproductive stages; therefore, it provides an index for assessment of health of the individual [7]. Copulation of female *C. pagurus* occurs between December and February when the females have moulted, allowing the males to mate successfully over three to twenty-one days [7]. Oviposition occurs four months post copulation, typically between January and June, with egg brooding continuing into the next eight months [7]. There have been no comparative studies investigating the SOM (utilising gonadal maturity or sperm plugs) of *C. pagurus* in the Northumberland and Berwickshire regions. It is essential to understand at what stage crabs reach sexual maturity at a regional scale to properly inform sustainable fisheries management and allow *C. pagurus* the opportunity to reproduce at least once before being landed, thus safeguarding and prolonging the fishery for future generations, providing long-term benefits to local, and national economies.

Table 2. Previous size at onset of sexual maturity (SOM) literature for *C. pagurus* within the UK.

Region	Sex	Maturity Metric	Size at 50% Maturity (CW_{50}) (mm)	Reference
North Wales	M	Gonad development	56–94	[7]
North Wales	F	Gonad development	86–105	[7]
South Wales	M	Gonad development	56–94	[7]
South Wales	F	Gonad development	101–115	[7]
Norfolk England	M		105	[9]
Norfolk England	F		110	[9]
Selsey England	M		115	[9]
Selsey England	F		125	[9]
E Coast England	F	Sperm plugs	116	[10]
SW Ireland	F	Mature gonads	127–139	[10]
England	M	Chelae	110	[10]
Shetland Scotland	M	Averaged all	116	[11]
Shetland Scotland	F	Averaged all	128	[11]
Shetland Scotland	F	Sperm plugs	123	[11]
Shetland Scotland	M	Mature gonads	125	[11]
Shetland Scotland	F	Mature gonads	133.5	[11]
Shetland Scotland	F	Hatched	144	[11]
Shetland Scotland	M	Averaged functional	125	[11]
Shetland Scotland	F	Averaged functional	139	[11]
Western Channel England	F	Mature gonads	137–147	[12]
Scotland East and West	M	Mature gonads	101–106	[13]
Scotland East and West	F	Mature gonads	127–128	[13]
Scotland	F	Sperm plug	122.9	[14]
Scotland	F	Ovigerous	143.7	[14]
Ireland	F	Mature gonads	132–138	[15]
Eastern Channel England	M	Mature gonads	105	[16]
Eastern Channel England	F	Mature gonads	126	[16]
Western Channel England	M	Mature gonads	90	[16]
Western Channel England	F	Mature gonads	112	[16]
North Sea	M	Mature gonads	89	[16]
North Sea	F	Mature gonads	109	[16]
Ireland	F	Gonad development	120	[17]
Isle of Man, Irish Sea	M	Gonad maturity	89	[18]
Isle of Man, Irish Sea	F	Gonad maturity	108	[18]

Note: CW_{50} represents the carapace width (mm) at which 50% of the population are deemed sexually mature.

SOM can be used to estimate the reproductive output of individuals before they are caught according to the current MCRS. Here, we conduct a cross-border fisheries project to determine whether the SOM for local *C. pagurus* populations (males and females) in both Berwickshire and Northumberland regions are different and assess whether the current MCRSs are suitable for each region.

2. Materials and Methods

C. pagurus individuals were collected from the Berwickshire region by local inshore fishermen over three fishing trips. Derogations were granted by Marine Scotland over the sampling period to allow the landing of individuals below the Scottish MCRS of 150 mm. The surveys were in accordance to terms of Section 9 of the Sea Fish Conservation Act 1967, Article 25 of Council Regulation No. 2019/1241, the specified crustaceans (prohibition on landing, sale and carriage) (Scotland): order 2017 No. 455 and the undersized edible crabs (Scotland) order 200 No. 228. In the Northumberland region (River Tyne to the Scotland/England Border) inshore creel fleets (eight fishing trips) deployed by the Northumberland Inshore Fisheries Conservation Authority (NIFCA) and local fishing vessels in Northumberland (two fishing trips) were used for sample collection (Figure 1).

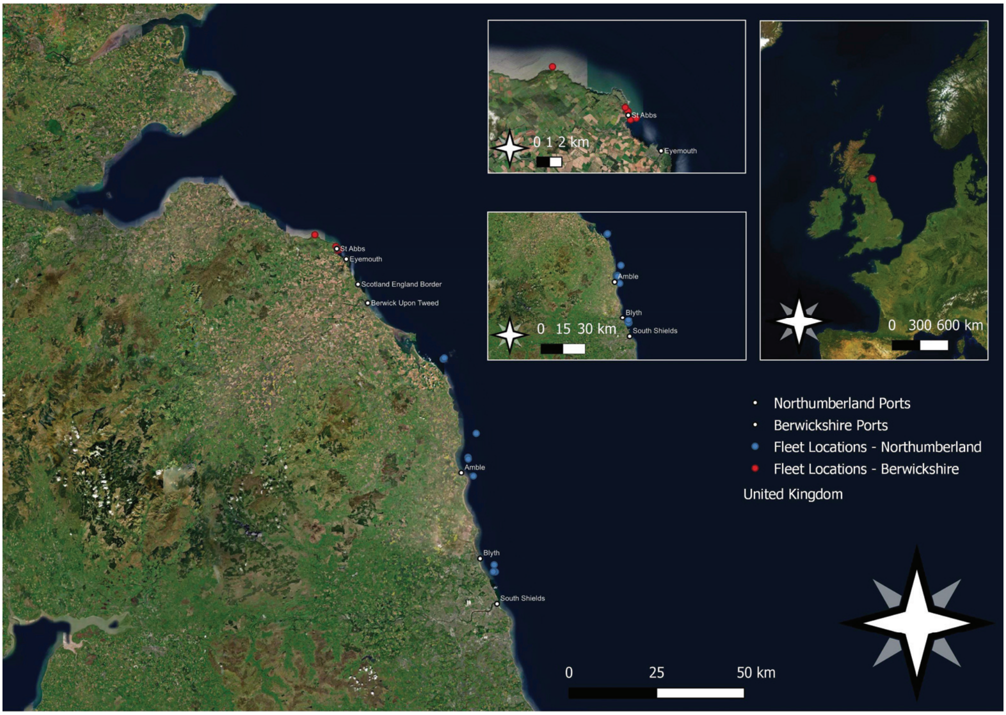

Figure 1. Map of locations from which the *C. pagurus* were sampled from both the Berwickshire (55.902769, −2.128988) and Northumberland (55.137154, −1.437651) areas of survey. Sampling locations for Northumberland are highlighted by the blue dots whereas the sample locations for Berwickshire are highlighted by the red dots.

A total of 768 individuals (carapace width range: 60–209 mm) were collected from both sites between September 2020 and May 2021 which were processed at St. Abbs Marine Station. Those *C. pagurus* individuals collected from the Berwickshire region were held in 3000–10,000 L aquarium tanks filled with ambient seawater (mean ± sem: 12.25 ± 0.46 °C, 33.84 ± 0.13 ppt). Those collected from the Northumberland district were frozen prior to arrival to the marine station due to storage and transportation difficulties. These samples were defrosted over a 24 h period at room temperature (22 °C) to allow complete thawing of the organ tissue. Thawing of the organ tissue allowed for the organs and tissue to be removed more easily, limiting the chance of wasted target tissue.

Initially, each fresh individual from Berwickshire was subject to a cold shock treatment, whereby the individual was placed in a freezer for a duration of 30 to 60 min at a temperature range of −16 to −20 °C. This treatment allowed the individuals to be desensitised before the dissections took place. To ensure the individual was dead before dissection, it was monitored for 5 min during which time any movement from chelae, pleopods and eye stalks was associated with inadequate desensitising and the individual was placed back in the freezer for an additional 30 to 60 min. Before each dissection took place, morphological data were collected from each individual using measurement callipers (Proops 350 mm Inside Outside Plastic Calliper Metric Measuring Scale) and a calibrated AMIR™ digital scale. For each individual, the following were recorded: wet weight (g), carapace width (mm), chelae length (mm), chelae height (mm), chelae depth (mm), sex, condition index, and moult stage. Morphological measurements of the chelae were assigned to the right claw. If the individual had a regrowing right chela, or had the right chela missing, the left chela was used for these measurements as the chelae are not dimorphic [7,19]. The condition index is a method of determining the general condition of an individual's morphometry based on criteria used in [7] (Table 3). The extent of black spot coverage on the body was also noted for each individual based on criteria from [7,10,20]. The moult stage was determined using the same table of criteria used in the paper [7] (Table 4).

Table 3. Descriptions detailing the criteria to meet each condition index stage.

Condition Index	Condition Description
1	High standard of health. Chelipeds present. Chelae present. No black spot lesions. No damage.
2	Good standard of health. One to two chelipeds are missing. Limited black spot lesions.
3	Average standard of health. More than two chelipeds missing. Black spot lesions present.
4	Bad standard of health. One or both chelae missing. Black spot lesions covering around 50% of the carapace. Limited damage to the body.
5	Poor standard of health. One or both chelae missing and chelipeds missing. Large surface area of black spot lesions on the body. Damaged carapace.

Note: The information in this table is developed from [7,10,20].

Table 4. Visual descriptions of moult stages to assign to each individual *C. pagurus*.

Moult Stage	Description
Early Post Moult	Soft, white, no biofouling on carapace, very sharp toes.
Recent Moult	No biofouling on carapace, sharp toes, carapace not fully hardened.
Inter-Moult	Carapace covered with biofouling usually in the form of biofilm attached to the hairs of the chelipeds, toes are worn to a smooth rounded shape.
Degraded	Carapace shows signs of great biofouling in the form of tubeworms, barnacles, large surface area coverage of biofilms especially attached to the hairs of the chelipeds, damage to carapace (holes, indentations), distinguishable smell to the individual.

Note: Details and descriptions used to assign the stages were based on those from [7,21].

Females were observed for sperm plugs prior to dissection, which were noted if present. The crab dissection took place on the ventral side of the animal. An incision at the telson allowed for the dissection to take place along the moult line. This allowed the dorsal and ventral sides to be separated, opening the body cavity. Once the body cavity was exposed, the hepatopancreas, which sits on top of the gonads, was removed for weighing post dissection. The gonads were then exposed and a photo was taken for post sampling analysis. The sex stage was then determined by comparing the photo with information and details from the literature [7,10,19,20]. An example of the information and details is summarised in Table 5, Figure 2.

Table 5. Visual descriptions used to assign the gonad stages for each *C. pagurus* individual.

Female					
Stage	1-Immature	2-Undeveloped	3-Developing	4-Mature	5-Recovery
Details	No egg cells present	Pre-vitellogenesis	Early secondary vitellogenesis	Late secondary vitellogenesis	Post Reproductive
Visual	Gonad is transparent and thin in structure.	Gonad lobes are visible with a light pink/ grey coloration.	Gonad has more noticeable pink colour. Covers less than 50% of the cavity.	Ovaries are very large, covers over 50% of the cavity with a prominent orange/red colour.	Remnant eggs are visible with the ovary exhibiting a loose structure and white appearance.

Table 5. *Cont.*

Male			
Stage	1-Immature	2-Developing	3-Mature
Details	Spermatids	Spermatozoa	Spermatophore
Visual	Small testes that are transparent or undetectable.	White and obvious testes.	Both swollen testes and vas deferens.

Note: Descriptions are based on those by [7,8,10,19,21].

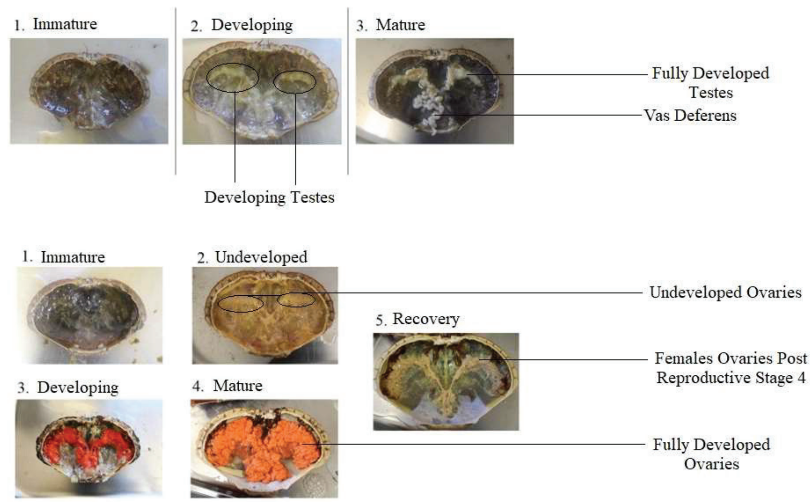

Figure 2. Visual representations of the three gonad stages of male *C. pagurus* and five gonad stages of female *C. pagurus*. For males, stages 1 to 3 are presented from left to right at the top of the image. For females, stages 1 and 2 are presented left to right on the top, stages 3 and 4 are presented left to right on the bottom. Stage 5 is shown on the far right.

The wet weight of the hepatopancreas was measured using a calibrated Sartorius™ AC 211S-00MS Iso Cal digital scale. If the gonad stage for males were three, and three or four for females, deeming the individual sexually mature, the gonad was removed and the wet weight recorded to the same criteria as the hepatopancreas. The hepatosomatic index (HSI), a means of indicating lipid stores in the individual, is calculated by the hepatopancreas wet weight (HWW) (g) divided by the total wet weight (WW) (g) of the individual to give a percentage value (HSI).

$$HSI = HWW/WW \times 100$$

Similar to the hepatosomatic index, the gonadosomatic index (GSI) is calculated by the gonad wet weight (GWW) (g) divided by the total wet weight (WW) (g) to provide a percentage value.

$$GSI = GWW/WW \times 100$$

2.1. Statistical Analysis

All statistical analysis was performed in R Studio [22]. Initial data were analysed for normal distribution and homogeneity of variance using the Shapiro–Wilk test, normality histograms, and Levene's Test. Normality and homogeneity of variance was considered when significance was interpreted as p-value > 0.05. Size at gonadal and morphometric maturity of the individuals was estimated using the sizeMat package (version 1.1.2, published: 2 June 2020) [23]. Segmented regression analysis and models were conducted using the segmented package (version 1.3.4, published: 22 April 2021) in R Studio. Data were separated by location (Berwickshire and Northumberland) to allow for regional comparisons. Comparisons made regarding the hepatosomatic index (HSI) were analysed using Anova tests partnered with post hoc Tukey HSD tests. The gonadosomatic index (GSI) was tested with sex, gonad stage, and condition index using linear regressions. All statistics were tested to the significance value of 0.05.

2.2. Morphometric Maturity

Individuals were assigned into two groups (immature = 0 and mature = 1) which were classed using the allometric measurements (frequentist logit approach), X = independent variable (carapace width) and Y = dependent variable (chelae height, depth, and length) [24]. CW_{50} is the value given when there is a 50% chance at a given carapace width (mm) the individual is considered mature [25,26]. In the regression analysis, X is considered the explanatory variable, in this study carapace width (mm), and the classification of maturity CS (juveniles: 0, adults: 1) is considered the response variable (binomial). The variables are fitted to a logit function [24]:

$$PCS = 11 + e^{-(\beta^0 + \beta^1 x)}$$

where PCS is the likelihood of an individual being mature at carapace width (x). β^0 (intercept) and β^1 (slope) are parameters estimated. The CW_{50} is calculated as:

$$CW_{50} = -\beta^0 \beta^1$$

Maturity ogives were presented as graphs which highlighted the CW_{50} for each location and sex. Segmented regression was applied to assess breakpoints (BP) and confidence intervals at which morphometric data collected (chelae depth, chelae height and chelae length) indicated morphometric maturity in relation to allometric relationships [23]. The segmented regression follows the process of measuring the distance between two fitted lines at each respective breakpoint using the minimisation of a parameter [23].

2.3. Physiological Maturity

The size at physiological maturity is assessed using the carapace width (mm) in relation to the maturity stage assigned during the histological dissection process. The function follows a logistic approach in which the logit regression is based on a general linear model (GLM) [23]. As per the morphometric maturity, the function requires an allometric variable (X), in this case carapace width (mm) and stage of sexual maturity (immature = 0, mature = 1). Similar to the morphometric ogives, the gonad maturity ogives were presented as the fitted values as a curve logistic regression with confidence intervals (95%). Also highlighted is the CW_{50} for each location and sex.

3. Results

From both regions of sampling (Berwickshire and Northumberland), 768 individual *C. pagurus* were collected and dissected between the period of September 2020 and May 2021. A total of 501 males and 267 females were collected with 283 (56.49%) of these individuals considered mature (gonad stage three for males; three and four for females). Across the sampled population used in this study, 78.45% (n = 222) of the cohort were categorised as mature from the Berwickshire group and 21.55% (n = 61) from the Northumberland group. The smallest carapace width (mm) recorded was 60 mm and the largest being 209 mm (Figure 3. The smallest recorded wet weight (g) was 25.2 g and largest being 1345 g (Figure 4). Location-specific results showed that of those collected in Berwickshire, 222 were considered mature (190 males, 32 females) out of 437 individuals (313 male, 124 female). Morphological measurements showed a carapace width range of 72–209 mm and wet weight range of 50–1345 g. Those collected in Northumberland, 61 were considered mature (56 males, 5 females) out of 332 individuals (88 males, 144 females). Morphological measurements showed a carapace width range of 60–186 mm and a wet weight range of 25.2–810.6 g. All females were assessed for the presence of sperm plugs, nine females in

Berwickshire and 25 females in Northumberland presented sperm plugs. Black spot was also recorded as a measure of condition for the health of the individuals sampled. A total of 31 individuals in Berwickshire and 11 individuals from Northumberland were shown to have black spot present. Out of all individuals collected and sampled, 76 had their left claw measured due to missing or re-growing right chela. Of these 76 individuals, 49 were collected from Northumberland (males = 32, females = 17) and 27 from Berwickshire (males = 19, females = 8). Condition of the individuals did not greatly differ between the regions of sampling, those sampled from Berwickshire; males averaged 2.14 ± 0.06 and females 2.21 ± 0.10. Those sampled from Northumberland showed the greater condition indices on average, with males 3.17 ± 0.08 and females 2.78 ± 0.10, showing worse conditioned individuals. A full summary of the measurements taken is presented in Table 6.

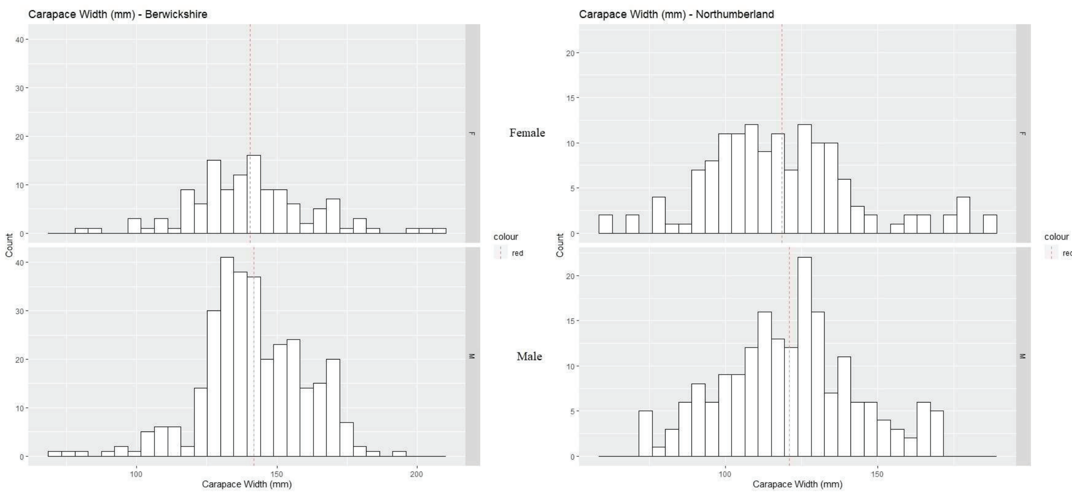

Figure 3. Carapace width (mm) of the Berwickshire (n = 437; (Males = 313, Females = 124)) and Northumberland (n = 332; (Males = 188, Females = 144)) *C. pagurus* sampled cohorts. The red dotted line highlights the mean carapace width (mm) for each population separated by sex.

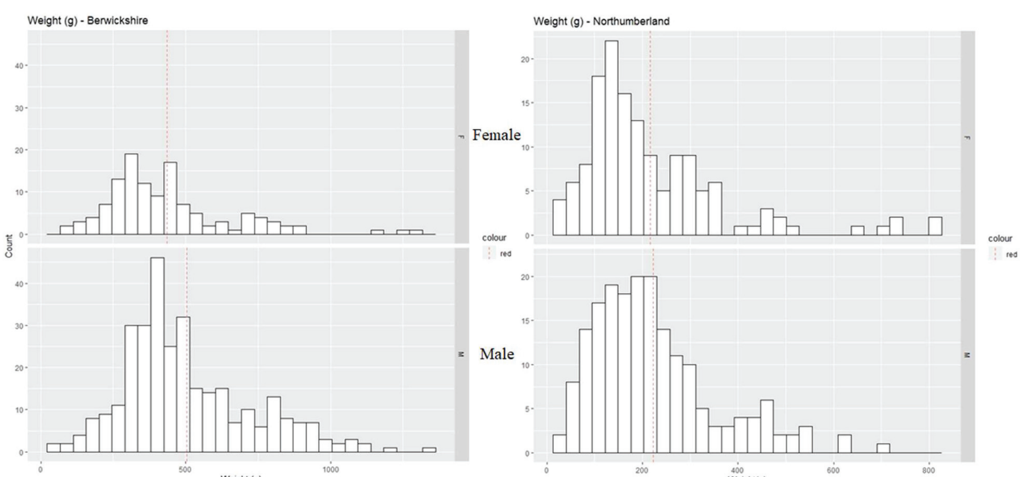

Figure 4. Wet weight (g) of the Berwickshire ((n = 437); (Males = 313, Females = 124)) and Northumberland ((n = 332); Males = 188, Females = 144) *C. pagurus* sampled cohorts. The red dotted line highlights the mean wet weight (g) for each population separated by sex.

Table 6. Mean and standard error of the morphometric measurements taken during the pre-dissection stage of each *C. pagurus* individual from both regions.

Region	Sex	CW (mm)	WW (g)	CI	MS	CL (mm)	CD (mm)	CH (mm)	HW(g)	GS
Berwickshire	M	141.76 ± 1.05	503.52 ± 12.60	2.14 ± 0.06	3.16 ± 0.03	34.51 ± 0.39	23.22 ± 0.29	37.18 ± 0.43	33.44 ± 0.77	2.45 ± 0.04
	F	140.48 ± 1.99	436.33 ± 19.93	2.21 ± 0.10	3.11 ± 0.07	27.57 ± 0.46	18.18 ± 0.29	29.50 ± 0.46	69.55 ± 34.72	2.29 ± 0.11
Northumberland	M	121.03 ± 1.63	222.89 ± 9.40	3.17 ± 0.09	2.89 ± 0.05	26.75 ± 0.55	17.55 ± 0.40	29.04 ± 0.63	11.65 ± 0.70	1.99 ± 0.06
	F	118.56 ± 2.07	216.34 ± 12.65	2.78 ± 0.11	2.70 ± 0.06	23.20 ± 0.46	14.77 ± 0.30	24.83 ± 0.48	11.59 ± 0.90	1.93 ± 0.12

Note: All data are presented as mean ± sem. Abbreviations in the table are as follows: CW—carapace width, WW—wet weight, CI—condition index, MS—moult stage, CL—chelae length, CD—chelae depth, CH—chelae height, HW—hepatopancreas weight, and GS—gonad stage.

3.1. Physiological Maturity

Physiological maturity (CW_{50}) was considered using the maturity stages three for males and three–four for females. Following the logistic regression with bootstrapping, it was considered that the size at which gonadal maturity was met for 50% of the population to be 108.5 mm (95% CI, 97.7–116.4 mm) for males in Berwickshire and 109.9 mm (95% CI, 105.9–113.5 mm) for those in Northumberland (Figure 5. For females, this was 126.8 mm (95% CI, 122.2–130.9 mm) in Berwickshire and 120.8 mm (95% CI, 117.2–125.3 mm) in Northumberland (Figure 5). In both regions, the females showed maturity at larger carapace widths when compared to males. Considering the ogive results, the *C. pagurus* in the Northumberland district would be mature at the current MCRS of 130 mm, and the same for the *C. pagurus* cohort from Berwickshire under the current Scottish MCRS of 150 mm.

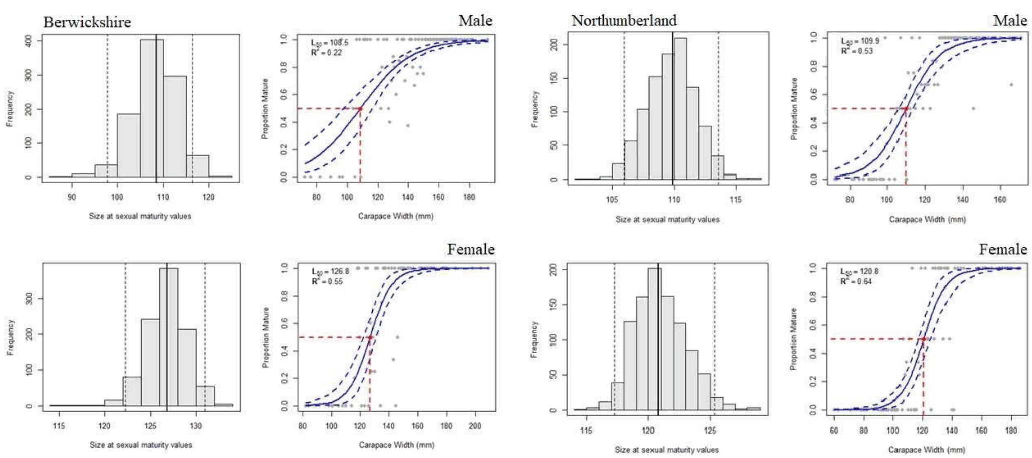

Figure 5. Male and female gonad maturity of the *C. pagurus* cohorts sampled from Berwickshire and Northumberland. The point at which 50% of the population are said to be mature is highlighted in red (CW_{50}), with confidence intervals (95% CI) shown by the blue dashed lines.

3.2. Morphometric Maturity

Segmented regression was used to calculate the carapace width (mm) in which there were changes in allometry relationships as described in [23]. The parameters used chelae length (mm), chelae depth (mm), and chelae height (mm) in regard to male *C. pagurus*. Outputs from the segmented regression for the male individuals in Berwickshire showed the chelae length break point (BP) at 119.07 ± 12.37 mm (mean ± sem), chelae depth breakpoint (BP) at 83.99 ± 7.33 mm (mean ± sem), and chelae height breakpoint (BP) at 137.99 ± 9.77 mm (mean ± sem). Outputs from the segmented regression for the male individuals in Northumberland showed the chelae length break point (BP) at 129 ± 6.72 mm (mean ± sem), chelae depth breakpoint (BP) at 119.67 ± 5.68 mm (mean ± sem), and chelae height breakpoint (BP) at 129 ± 4.77 mm (mean ± sem).

Using the sizeMat package in R Studio to determine the morphometric maturity of the *C. pagurus* individuals sampled in both regions, the relationship (linear regression) between carapace width (mm) and chelae height (mm) was tested. This method allowed for morphometric maturity analysis of females. In Berwickshire, the carapace width at which morphometric maturity was met for males was 141.1 mm (R^2 = 0.7, CI = 139.4–143 mm) and 134.7 mm (R^2 = 0.89, CI = 132.8–136.8 mm) for females. In Northumberland, carapace width at which morphometric maturity was met was 130.1 mm (R^2 = 0.84, CI = 128–132.5 mm) for males and 120.8 mm (R^2 = 0.64, CI = 117.2–125.3 mm) for females. A summary of the results from both regression analyses used can be seen in Table 7.

Table 7. Summary of outputs from the regression analysis (segmented and linear) considering the relationship between carapace width (mm) and other morphometric measurements (chelae height (CH), chelae depth (CD) and chelae length (CL) (mm)) from sampled individuals of *C. pagurus* from Berwickshire and Northumberland.

Region	Measurement (mm)	Sex	N	Segmented Regression			Linear Regression		
				Slopes	BP (mm)	R^2	R^2	BP (mm)	CI
Berwickshire	CH	M	308	0.187, 0.233	137.99 ± 9.77	0.85	0.7	141.1	139.4–143
	CD	M	308	0.108, 0.125	83.99 ± 7.33	0.76	0.83	142.3	140.9–143.7
	CL	M	308	0.166, 0.225	119.06 ± 12.37	0.86	0.86	153.3	151.8–154.9
	CH	F					0.89	134.7	132.8–136.8
Northumberland	CH	M	170	0.178, 0.266	129 ± 4.77	0.92	0.84	130.1	128–132.5
	CD	M	170	0.105, 0.155	119.67 ± 5.68	0.89	0.92	121.8	120.4–123.3
	CL	M	170	0.012, 0.015	129 ± 6.72	0.89	0.93	121.2	119.9–122.5
	CH	F					0.64	120.8	117.2–125.3

Note: The segmented regression is associated with the morphometric maturity and the linear regression is associated with the gonad maturity. BP is the estimated carapace width breakpoint.

3.3. Hepatopancreas Weight and Hepatosomatic Index (HSI)

In Berwickshire (Figure 6), the hepatopancreas wet weight was significantly different dependent on the gonad maturity stage of both sexes (F-value = 5.59, $p < 0.05$), in which gonad maturity stages two, three and four ($p < 0.05$) were different to gonad stage one. Hepatopancreas weight for males with gonad stages one, two and three ($p < 0.05$) was significantly different to females of gonad stage two. In comparison, for both sexes in Northumberland (F-value = 31.82, $p < 0.05$) (Figure 6) their hepatopancreas wet weight was significantly different between gonad stage one and all other sex-specific gonad stages ($p < 0.05$). Gonad stage four in females was significantly different to gonad stages two, three and five ($p < 0.05$). Variations in hepatopancreas weight were found between the sexes at different *C. pagurus* gonad stages in Northumberland. Gonad stage one in males was significantly different to gonad stage two in females ($p = < 0.05$). Male gonad stages one, two and three ($p < 0.05$) were significantly different to female gonad stage four. Only gonad stages one ($p < 0.05$) and two ($p = 0.01$) in males showed significant difference from female gonad stage five.

The HSI values for the Berwickshire-sampled individuals showed significant difference by gonad maturity stage (F-value = 61.83, $p < 0.05$), sex (F-value = 23.93, $p < 0.05$), and gonad stage as a covariate with sex (F-value = 11.78, $p < 0.05$). Contrastingly, the HSI for the individuals sampled from Northumberland showed no significance for gonad stage (F-value = 0.25, $p = 0.90$), sex (F-value = 0.78, $p = > 0.05$), or as covariates (F-value = 0.02, $p = 0.97$). Further results indicated that the HSI of the gonad stages one and two are significantly different to all other gonad stages ($p < 0.05$). There was a significant difference between gonad stages five and three ($p = 0.01$). Differences found between covariates indicated that gonad stages one and two in males showed significant difference to the female gonad stages two, four, and five ($p < 0.05$). Male gonad stage three showed significant difference from female gonad stage two ($p < 0.05$). The condition of the individual was shown to affect the HSI of the individuals (Anova, F-value = 5.80, $p < 0.05$), in particular that condition stage three was significantly different to stages one and two (Tukey HSD, $p < 0.05$). As covariates, condition and sex did not seem to affect the HSI value (Anova, F-value = 1.12, $p = 0.34$). This considers those sampled from Berwickshire (Figure 7), as for those in Northumberland (Figure 7) the opposite occurred, as covariates condition and sex affected the HSI significantly (Anova, F-value = 6.00, $p < 0.05$). Condition stage one in males was significantly different for condition stages four and five (Tukey HSD, $p < 0.05$). Condition stage four was significantly different to stage two (Tukey HSD, $p < 0.05$) for males. The HSI values were also significantly different between the first condition stages in males and females in Northumberland (Tukey HSD, $p = 0.02$). As separate factors, condition (Anova, F-value = 1.62, $p = 0.16$) and sex (Anova, F-value = 0.32, $p = 0.57$) did not affect HSI in Northumberland.

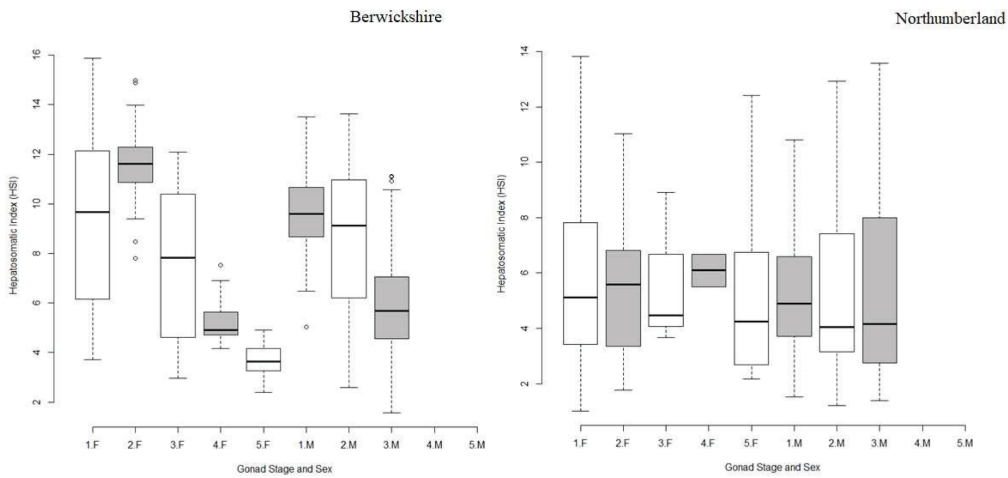

Figure 6. Hepatosomatic index (HSI) of both sexes considering all gonad maturity stages from individuals sampled in Berwickshire and Northumberland. Data are presented as mean ± sem. Circles indicated in the image present outliers post data analysis.

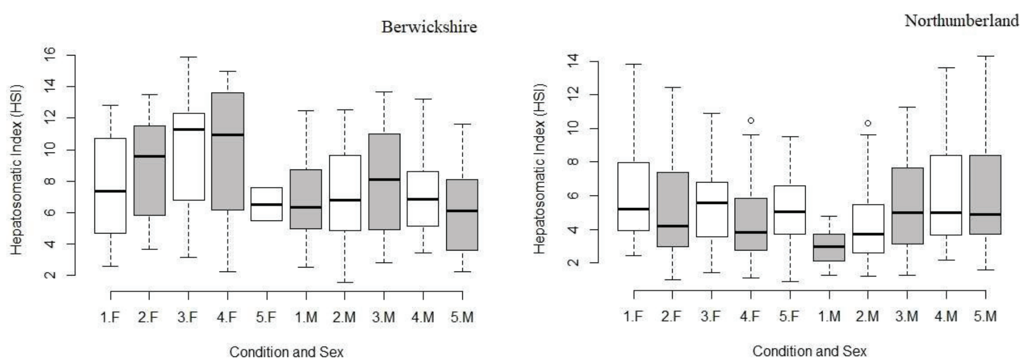

Figure 7. Hepatosomatic index (HSI) based on the condition and sex of the individuals sampled from Berwickshire and Northumberland. Data are presented as the mean ± sem. Circles indicated in the image present outliers post data analysis.

3.4. Gonad Weight and Gonadosomatic Index (GSI)

The GSI values for the sampled individuals were separated by region of sampling. For those sampled from Berwickshire, the GSI was significantly different regarding sex (F-value–56.76, $p < 0.05$) and gonad stage (F-value = 40.99, $p < 0.05$) following linear regression (R^2 = 0.42, p < 0.05). For those individuals from Northumberland, the GSI was significantly different for sex (F-value = 25.74, p < 0.05), gonad stage (F-value = 3.38, p = 0.04), and condition index (F-value = 5.23, p = 0.02), following linear regression (R^2 = 0.35, p < 0.05). The GSI for males showed no great difference between the condition indices in Berwickshire, but showed a relatively incremental change with condition indices in Northumberland (Table 8). The GSI of the females showed the greatest value during condition index two in both regions (Table 8).

Table 8. The gonadosomatic index for each condition index based on sex of the individuals.

Sex	Condition Index	Gonadosomatic Index (GSI)	
		Berwickshire	**Northumberland**
M	1	2.32 ± 0.09	1.60 ± 0.51
	2	2.15 ± 0.10	1.28 ± 0.16
	3	1.95 ± 0.21	2.07 ± 0.37
	4	2.80 ± 0.21	2.61 ± 0.31
	5	2.87 ± 0.19	2.65 ± 0.30
F	1	4.86 ± 0.87	5.11 ± 1.86
	2	7.39 ± 1.29	6.03 ± 3.05
	3	1.37 ± 0.41	2.34
	4	4.06 ± 0.83	NA
	5	4.40	NA

Note: The table presents the gonadosomatic index for the sampled individuals from Berwickshire and Northumberland as mean ± sem. No individuals collected from Northumberland were categorised as condition indexes 4 and 5. Only one female individual was category 5 from the Berwickshire cohort and category 3 from Northumberland. NA indicated in the table highlights non applicable values as no sampled female individuals (n = 0) from Northumberland were considered condition index value 4 or 5.

4. Discussion

4.1. Physiological Maturity

The size at onset of sexual maturity based on the gonad characteristics used in this study highlights a difference between the study regions of Berwickshire and Northumberland (Figures 8 and 9). Across the sampled population, 78.45% of the cohort were categorised as physiologically mature from the Berwickshire group and 21.55% from the Northumberland group. Previous studies have shown that *C. pagurus* with carapace widths of <100 mm can equate to 38% of the sampled cohort of which 25% females and 50% of males were deemed physiologically mature; this was not found for individuals in Berwickshire or Northumberland [7,26]. Males showed a 1.4 mm CW_{50} difference between Berwickshire and Northumberland, with those exhibiting a greater CW_{50} in Northumberland. Males are considered to reach sexual maturity at smaller sizes than females [7,27]. Contrastingly, females in Northumberland expressed a lower CW_{50} than those in Berwickshire with a CW_{50} difference of 6 mm between the regions. The ability for males to mature at smaller sizes could be of benefit by improving the probability of mating success in populations where the ratio is in favour of females and competition is higher. Discrepancies between regions in relation to sexual maturity is common, as previous studies have shown that males can express a CW_{50} range of 56–125 mm and females 86–133.5 mm across the UK, and in a recent study by [8] the CW_{50} for East Scotland was 101–106 mm for males and 127–128 mm for females which coincides with the results found in the literature [8,11,28]. Previous reports from stock assessment surveys by NIFCA suggested a CW_{50} of 89.5 mm for males and 111.6 mm for females [29]. The female CW_{50} value from this assessment follows suit to the value of [28] at 112 m; however, from this study there is a 8–14 mm difference when both locations are considered which suggests a regional variation across a latitudinal range. It should be stated that the study by [29] used parameters specified by CEFAS (The Centre for Environment, Fisheries and Aquaculture Science), which are used for their assessments, to inform stock assessments rather than specific SOM. In this study, an increase of 20.4 mm for males and 9.4 mm for females over the three-year period was reported (2019–2021). This increment in CW_{50} could be due to the smaller sample size (n = 332) compared to the stock assessment by NIFCA (n = 11811) [29]. The current MCRS values for these regions (150 mm for Scotland and 130 mm for England) are appropriate for each respective region, allowing 50% of the populations to reproduce before the probability of being harvested. It has been suggested that the regional variations could be due to the availability of sexual partners, population density, and environmental factors [7,8,10,12,19,30–32]. Environmental factors such as temperature and chemicals have been suggested as a determinate of ovary maturation, with warmer waters associated with lower SOM and chemicals associated with embryonic mortality and lower egg production [32–34]. Growth rates vary after the puberty moult whereby the females express more energy into reproduction than the males [9,31]. It is suggested that static gear contain disproportionately greater numbers of larger crabs than the use of trawling and other active mobile methods of fishing [7]. The period of sampling covered the months of September 2020 and April/May 2021. It has been suggested that the spawning period of *C. pagurus* is in the winter months [13] and more specifically between November and January [12,29], suggesting that the cohort selected for this study would have been in their final stages of reproduction. From all 768 individuals sampled, only two females bore eggs. Thus, no functional maturity conclusions could be suggested in this study. Ovigerous females are rare to catch using commercial fishing methods [19,35,36] which has been suggested due to nesting behaviours and burying in finer sediment offshore. It has been observed that the typical carapace width range for females bearing eggs is between 113 mm and 144 mm in Northern Europe with a minimum size of 129 mm associated with samples in Northern England [21] and with increasing size comes greater fecundity in this species [14,37]. Associating this minimum size from [37] to the female CW_{50} values in this study, it would be suggested there are mitigative measures to allow breeding before removal from the fishery. However, this only considers 50% of the population and there may be individuals removed at larger sizes bearing eggs which are regarded as the minimum size for the stock. Females were observed for presence of sperm plugs to which a total of 34 individuals (Berwickshire: 9; Northumberland: 25) presented one or two sperm plugs. The size of females presenting sperm plugs ranged from 112 to 186 mm in Northumberland and 118 to 170 mm in Berwickshire which is lower than the resultant ~80 mm shown in previous studies [8,17]. It is suggested that the presence of behavioural maturity is met at lower carapace widths than suggested from the gonadal maturity results in this study [8,17]. Females are considered fully mature when gonadal and morphometric maturity are met [8,17], such as the production and carrying of eggs [8,17,34]. Males are considered mature when copulation is successful [21]; however, in this study six individuals (Berwickshire: four (CW: 129–174 mm); Northumberland: two (CW: 129–166 mm)) only presented one teste (carapace width: 129–174 mm). It is unknown whether this affects the success rate for copulation in these males. The interpretation of these results should be that this is not representative of the whole *C. pagurus* population sampled from Northumberland and Berwickshire. Using commercial vessels which were utilised to collect undersized individuals primarily with NIFCA deployed creels collecting those from a broader size range. It could be suggested that this would skew the data, therefore influencing the percentage of those deemed mature. However, the size range as presented in Figures 3 and 4 shows that the size distributions in each region (Berwickshire: 72–209 mm; Northumberland: 60–186 mm) did not vary heavily and therefore could be said that the results are representative, which is supported further by [38] who sampled a wide size range of *C. pagurus* representative of the commercial fishery but also suggested that this be considered when interpreting the results.

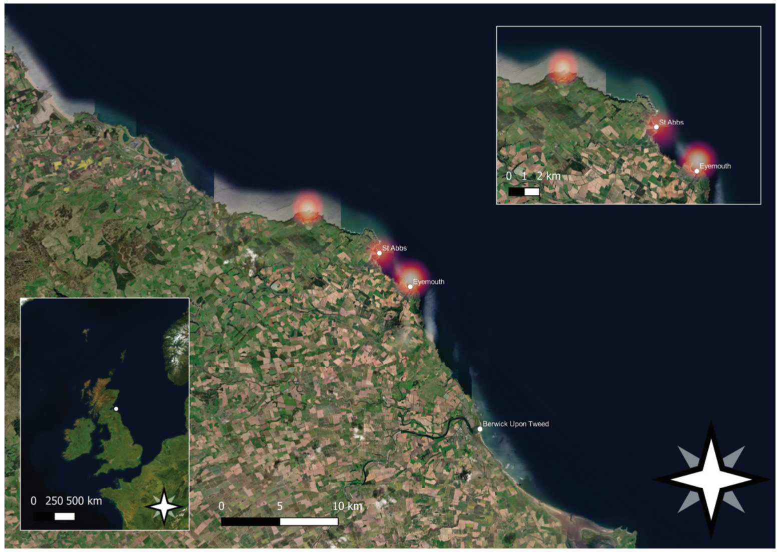

Figure 8. Heat map of the gonad maturity stages for both male (n = 313) and female (n = 124) *C. pagurus* individuals sampled in Berwickshire (55.902769, −2.128988). Gonad maturity stages three and four were considered as they correspond to sexual maturity for both sexes. The white colour of the map highlights the later maturity stage; in this case, gonad stage four, with gonad stage 3 emitting the pink/purple colour.

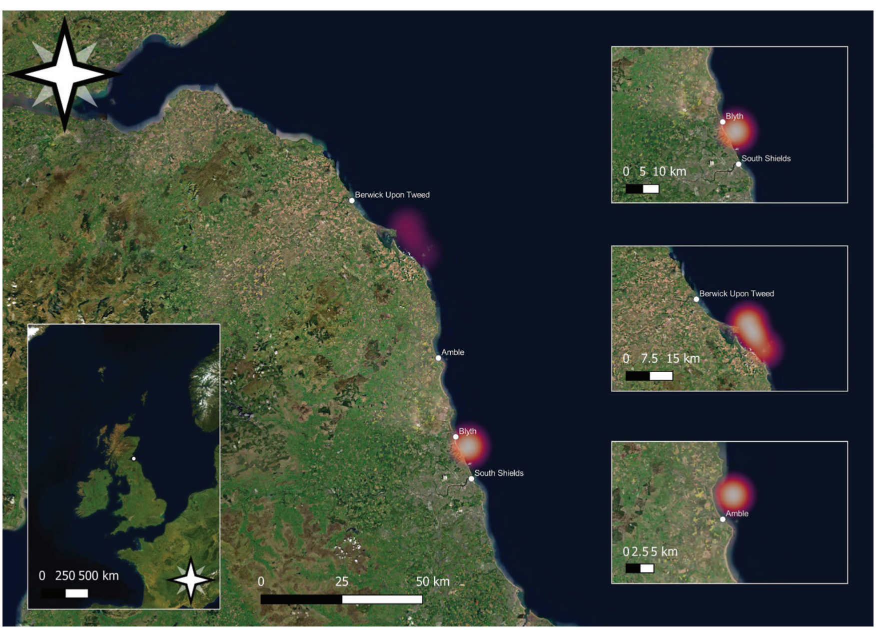

Figure 9. Heat map of the gonad maturity stages for both male (n = 188) and female (n = 144) *C. pagurus* individuals sampled in Northumberland (55.137154, −1.437651). Gonad maturity stages three and four were considered as they correspond to sexual maturity for both sexes. The white colour of the map highlights the later maturity stage; in this case, gonad stage four, with gonad stage three emitting the pink/purple colour.

4.2. Morphometric Maturity

Using morphometric measurements to determine onset of sexual maturity is commonplace in fisheries research [7,8,21,26,30,31,33]. When discussing the size at sexual maturity, morphometric data must be considered as an estimate due to the regional variations in individuals' growth rate and age at maturation [26]. The typical metric for males is the chelae and the abdominal flap widths for females as they indicate sexual dimorphism [7,26]; however, in the present study the same metrics were used across both sexes, the carapace width and chelae height, which followed the protocol of the sizeMat package. Behaviours indicative of courtship and combat signify a change in the allometry of male chelae, whereas a change in abdominal width for females relates to the accommodation of egg clutches [27]. It may be suggested that the nuance of sexual dimorphism in the females sampled may be lost due to the lack of abdominal flap measurements used in the analysis. The onset of sexual maturity based on the morphometrics on chelae length and carapace width was assessed by [31], whereby the allometry was met at 107 mm for males and 155 mm for females. Considering the results from the segmented regressions, males in Berwickshire (119.06 mm) and Northumberland (129 mm) show morphometric maturity at much greater sizes. Using morphometric measurements as the sole method of assessing maturity in this species provides inadequate results, as mentioned in [7,8], as mature males are underestimated, and mature females are overestimated. In this present study, the difference in the morphometric and gonadal maturity estimates varied. A difference of 32.6 mm for males, 7.9 mm for females in Berwickshire and a difference of 20.2 mm for males in Northumberland was found. Only the females from the Northumberland cohort showed no variation in estimates between the morphometric and gonadal CW_{50} values. A sex difference in maturation was stated by [15,26,27] in that males mature at smaller sizes than females which was stated to be documented by [10] also. This pattern of smaller maturation sizes in males compared to females was not found in both regions. A carapace width size difference of 6.4 mm in Berwickshire and 9.3 mm in Northumberland was observed, if morphometric maturity is considered solely. Sample size for the present study was much greater than that of [26], which could contribute to the contrast in patterns regarding smaller sizes of maturation in males. Fecundity of *C.pagurus* significantly increases with the size of the female carapace [14]. It has been recorded that age of maturity for female *C. pagurus* is four years which was the oldest age of maturation recorded by [14], and the reproductive cycle was predicted to be seven years which included an annual or two annual broods [14]. Fishing pressure is documented to change growth patterns over ecological and evolutionary time periods [7,28]. In this instance, ecological change by fishing pressure may suggest the change in SOM for the Northumberland cohort when NIFCA historic data in [33] are compared with this study. The NIFCA Byelaw: "Crustacea and Molluscs Permitting and Pot Limitation" states commercial fishermen in the district are limited to 800 fishing pots, whereas in the Berwickshire region there is no pot limitation. The fishing pressure in each region greatly varies and therefore regional variances in the size at onset of maturity would be expected, considering the suggestion by [7]. By applying movement tracking to the wider *C. pagurus* population, the ability to highlight whether these populations are indicatively different would show whether these populations are subject to variations in fishing pressure as we cannot suggest whether mature individuals migrate to and from these regions and thus affect the general SOM in each respective region.

4.3. Hepatopancreas Weight and Hepatopancreas Index (HSI)

It has been suggested that the HSI values in edible crab follow a capitalist strategy [7,21]. This suggests that the energy used in reproduction is not recovered post mating. HSI is a means of understanding the stored fats in the individual [7,21], as well as the reproductive cycle of gamete production in both sexes [7,16,21,26]. In this study, it could be perceived that this trend of energy loss during reproduction and lack of recovery post mating is found in the sampled population from Berwickshire but not for those from Northumberland (Figure 6). Not only from reproduction, but other factors could suggest the variations in HSI values such as the redirection of energy to chelae and limb regrowth and the negative effects of black spot disease whereby an energy deficiency is met when the health of the individual worsens, influencing the individual to divert energy stores towards reproduction [17,37]. Further investigation into factors, such as sea temperature and diseases, that could influence the HSI values of this species is required.

4.4. Gonad Weight and Gonadosomatic Index (GSI)

The GSI is determined as the means of measuring the reproductive timing of the individual as well as the spawning season for a group [39]. It was found that the GSI was significant to sex, gonad stage, and condition of the individual (Table 8). Similar results were found in [7]. Only in Berwickshire, the GSI was not significant with condition. The GSI generally increases with gonad stages following the maturation cycle of the individuals [40]. This has been observed by [16], and also in the present study (Table 8). GSI has also been shown to increase with black spot disease and in individuals considered to be in poor health [17,18,26], suggesting that in times of physical stress the individual will redirect energy into reproduction but there is no evidence of such behaviour.

5. Conclusions

This study highlights the variations between regions regarding the size at which sexual maturity is reached in the commercially important *C. pagurus*. The SOM (108.5–126.8 mm) found in this study allows for the maturing population to reproduce at least once before they are fished from this stock when related to the specific MCRS standards in both regions (150 mm Berwickshire and 130 mm Northumberland). It would be suggested then that the current MCRS values are effective at maintaining a viable brood stock for both regions. However, the variation in these regions still warrants the need for localised and regional monitoring of this species. A consistent monitoring procedure would be of benefit as anecdotal evidence suggests that fishermen are landing the 130 mm-sized individuals in Northumberland to counter the 150 mm legal limit in Berwickshire. NIFCA, through their monitoring activities, have a greater understanding of their fishery than would be stated for Berwickshire. Monitoring in Scotland is conducted over a larger scale which limits the nuances that are apparent in smaller, more localised, marine areas and therefore this study provides information that is specific and not generalised to Berwickshire. Further investigation into cross landing of sized individuals is required as removing individuals from the population that are not meeting the regional SOM may lead to a reduction in the stock. The data collected in this study provide a baseline for the region of Berwickshire, whilst providing additional information to the Northumberland Inshore Fisheries and Conservation Authority (NIFCA).

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Institutional Review Board Statement: The research adhered to the legal requirements of the country (U.K.) in which the work was carried out, and all institutional guidelines of St Abbs Marine Station. Informal ethical review and approval of this research was conducted by authors and trustees of St Abbs Marine Station using published literature and information obtained from Animals (Scientific Procedures) Act 1986 (ASPA). Animal collection was conducted with the local commercial fishing fleet. Derogations were granted by Marine Scotland over the sampling period to allow the landing of individuals below the Scottish MCRS of 150 mm. The surveys were in accordance to terms of Section 9 of the Sea Fish Conservation Act 1967, Article 25 of Council Regulation No. 2019/1241, the

specified crustaceans (prohibition on landing, sale and carriage) (Scotland): order 2017 No. 455 and the undersized edible crabs (Scotland) order 200 No. 228. At present crustaceans and cephalopods still remain outside the protections of the Animal Welfare Act, the Act which makes it an offence to cause unnecessary suffering to protected animals. The experiment does not need ethical approval.

Data Availability Statement: The data collected and analysed in this paper is not available publicly. Readers can inquire and request the raw data files from the first author (blair.easton@marinestation.co.uk).

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References

1. Seafish. Fleet Enquiry Tool. Available online: www.seafish.org (accessed on 25 September 2020).
2. M. Management Organisation. UK Sea Fisheries Statistics 2019. 2019. Available online: https://www.gov.uk/government/statistics/uk-sea-fisheries-annual-statistics-report-2019 (accessed on 25 September 2020).
3. The Scottish Government. The Specified Crustaceans (Prohibition on Landing, Sale and Carriage) (Scotland) Order 2017. Scotland. 2018. Available online: https://www.legislation.gov.uk/ssi/2017/455/contents/made (accessed on 25 September 2020).
4. Sea Fisheries, England Sea Fisheries, Northern Ireland Conservation of Sea Fish The Undersized Edible Crabs Order 2000. 2000. Available online: https://www.legislation.gov.uk/uksi/2000/2029?view=plain (accessed on 25 September 2020).
5. North Eastern Inshore Fisheries and Conservation Authority (NEIFCA). Minimum Conservation Reference Sizes. 2022. Available online: https://www.ne-ifca.gov.uk/minimum-conservation-reference-sizes (accessed on 6 May 2022).
6. The Scottish Government. Scottish Sea Fisheries Statistics 2020. 2020. Available online: https://www.gov.scot/publications/scottish-sea-fisheries-statistics-2020/ (accessed on 25 September 2020).
7. Haig, J.A.; Rayner, G.; Akritopoulou, E.; Kaiser, M.J. Fecundity of Cancer pagurus in Welsh waters; a comparison with published literature. *Fish. Conserv. Sci. Rep.* **2015**, 24.
8. Mesquita, C.; Dobby, H.; Sweeting, S.; Jones, C.S.; Pierce, G.J. Size-at-maturity of Brown Crab (*Cancer pagurus*) in Scottish waters based on gonadal and morphometric traits. *Fish. Res.* **2020**, *229*, 105610. [CrossRef]
9. Bennett, D.B. Factors in the life history of the edible crab (*Cancer pagurus* L.) that influence modelling and management. *ICES Mar. Sci. Symp.* **1995**, *199*, 89–98.
10. Edwards, E. *Edible Crab and Its Fishery in British Waters*; Fishing News Books: Farnham, UK, 1979.
11. Tallack, S.M.L. The Biology and Exploitation of Three Crab Species in the Shetland Islands, Scotland: Cancer pagurus, Necora puber and Carcinus maenas. Open University: Maidenhead, UK, 2002.
12. Latrouite, D.; Noel, P. Observations sur la Maturite Sexuelle et la Ponte du Tourteau Cancer Pagllrlls en Manche. 1993. Available online: https://archimer.ifremer.fr/doc/00000/1420/ (accessed on 25 September 2020).
13. Mesquita, C.; Dobby, H.; Pierce, G.J.; Jones, C.S.; Fernandes, P.G. Abundance and spatial distribution of brown crab (*Cancer pagurus*) from fishery-independent dredge and trawl surveys in the North Sea. *ICES J. Mar. Sci.* **2020**, *78*, 597–610. [CrossRef]
14. Tallack, S.M.L. The reproductive cycle and size at maturity observed in *Cancer pagurus* in the Shetland Islands, Scotland. *J. Mar. Biol. Assoc. United Kingd.* **2007**, *87*, 1181–1189. [CrossRef]
15. Andersens, H. Ices Scrab Report 2005 Ices Living Reources Committee Ices cm 2005/g:10 ref. d Report of the Study Group on the Biology and Life History of Crabs (Sgcrab) International Council for the Exploration of the Sea Conseil International Pour l'Exploration de la Mer. 2005. Available online: www.ices.dk (accessed on 19 September 2020).
16. Smith, M. Development of a Multiple Indicator Framework Macro-Crustacean Fishery Assessment and Management. 2010. Available online: https://secure.toolkitfiles.co.uk/clients/25364/sitedata/Redesign/Key_Species/Edible-Crab-Species-Profile.pdf?c (accessed on 19 September 2020).
17. Tully, O.; Robinson, M.; Cosgrove, R.; O'keeffe, E.; Doyle, O.; Lehane, B. The Brown Crab (*Cancer pagurus* L.) Fishery: Analysis of the Resource in 2004–2005. 2006. Available online: https://www.researchgate.net/publication/237612664_The_Brown_Crab_Cancer_pagurus_L_Fishery_Analysis_of_the_resource_in_2004-2005 (accessed on 15 April 2023).
18. Öndes, F.; Kaiser, M.J.; Murray, L.G. Relative growth and size at onset of sexual maturity of the brown crab, *Cancer pagurus* in the Isle of Man, Irish Sea. *Mar. Biol. Res.* **2017**, *13*, 237–245. [CrossRef]
19. Ungfors, A. Sexual maturity of the edible crab (*Cancer pagurus*) in the Skagerrak and the Kattegat, based on reproductive and morphometric characters. *ICES J. Mar. Sci.* **2007**, *64*, 318–327. [CrossRef]

20. Scott, K.; Harsanyi, P.; Lyndon, A.R. Understanding the effects of electromagnetic field emissions from Marine Renewable Energy Devices (MREDs) on the commercially important edible crab, *Cancer pagurus* (L.). *Mar. Pollut. Bull.* **2018**, *131*, 580–588. [CrossRef] [PubMed]
21. Haig, J.A.; Pantin, J.R.; Salomonsen, H.; Kaiser, M.J. *Size at Maturity of the Edible Crab (Cancer pagurus) in Welsh Waters*; Fisheries & Conservation Science report; Bangor University: Bangor, Wales, 2015; p. 26.
22. RStudio. *RStudio: Integrated Development for R. RStudio*; PBC: Boston, MA, USA, 2020; Available online: http://www.rstudio.com/ (accessed on 13 May 2021).
23. Torrejon-Magallanes, J. sizeMat: An R Package to Estimate Size at Sexual Maturity. 2020. Available online: https://cran.r-project.org/web/packages/sizeMat/vignettes/sizeMat.html#references (accessed on 13 May 2021).
24. Somerton, D.A. A Computer Technique for Estimating the Size of Sexual Maturity in Crabs. *Canadian J. Fish. Aquat. Sci.* **1980**, *37*, 1488–1494. [CrossRef]
25. Roa, R.; Ernst, B.; Tapia, F. Estimation of Size at Sexual Maturity: An Evaluation of Analytical and Resampling Procedures. 1999. Available online: https://spo.nmfs.noaa.gov/sites/default/files/13roafis.pdf (accessed on 6 May 2022).
26. Haig, J.A.; Bakke, S.; Bell, M.C.; Bloor, I.S.; Cohen, M.; Coleman, M.; Dignan, S.; Kaiser, M.J.; Pantin, J.R.; Roach, M.; et al. Reproductive traits and factors affecting the size at maturity of *Cancer pagurus* across Northern Europe. *ICES J. Mar. Sci.* **2016**, *73*, 2572–2585. [CrossRef]
27. Small, J.; Smith, C. *Edible/Brown Crab (Cancer pagurus)*; Southern Inshore Fisheries and Conservation Authority: Poole, UK, 2021.
28. Öndes, F.; Kaiser, M.J.; Murray, L.G. Torres, G. Reproductive Ecology, Fecundity, and Elemental Composition of Eggs in Brown Crab *Cancer pagurus* in the Isle of Man. *J. Shellfish. Res.* **2016**, *35*, 539–547. [CrossRef]
29. Boon, A. Stock Status Assessment for the *Cancer pagurus* Fishery of the Northumberland Coast in 2019. 2019. Available online: https://www.google.com/url?sa=i&rct=j&q=&esrc=s&source=web&cd=&cad=rja&uact=8&ved=0CAIQw7AJahcKEwjwpdfdk_D-AhUAAAAAHQAAAAAQAw&url=https%3A%2F%2Fnifca.gov.uk%2Fwp-content%2Fuploads%2F2022%2F02%2FFlag-Brown-Crab-Report-Final.docx&psig=AOvVaw20oS1v-d2m7Vma96TZj_Hu&ust=1683993752900926 (accessed on 25 September 2020).
30. Ungfors, A. Fisheries Biology of the Edible Crab (*Cancer pagurus*) in the Kattegat and the Skagerrak: Implications for Sustainable Management. Ph.D. Thesis, Göteborg University, Gothenburg, Sweden, 2008.
31. Le Foll, A. Contribution a l'etude de la biologie du crabe-tourteau Cancer pagurus sur les cotes de Bretagne sud. *Revue des Travaux de l'Institut des Pêches Maritimes* **1984**, *48*, 5–22.
32. Hartnoll, R.G. Growth, Sexual Maturity and Reproductive Output. In *Factors in Adult Growth*, 1st ed.; Wenner, A.M., Ed.; Routledge: Oxfordshire, UK, 1985. [CrossRef]
33. Bakke, S.; Larssen, W.E.; Woll, A.K.; Søvik, G.; Gundersen, A.C.; Hvingel, C.; Nilssen, E.M. Size at maturity and molting probability across latitude in female *Cancer pagurus*. *Fish. Res.* **2018**, *205*, 43–51. [CrossRef]
34. Shields, J.D. The reproductive ecology and fecundity of Cancer crabs. In *Crustacean Egg Production*; CRC Press: Boca Raton, FL, USA, 1991; pp. 193–213.
35. Hines, A.H. Fecundity and Reproductive Output in Nine Species of *Cancer* crabs (Crustacea, Brachyura, Cancridae). *Can. J. Fish. Aquat. Sci.* **1991**, *48*, 267–275. [CrossRef]
36. Hartnoll, R.G. Mating in the Brachyura. *Crustaceana* **1969**, *16*, 161–181. [CrossRef]
37. Hancock, D.A.; Edwards, E. Estimation of Annual Growth in the Edible Crab (*Cancer pagurus* L.). *ICES J. Mar. Sci.* **1967**, *31*, 246–264. [CrossRef]
38. Woll, A.K.; Berge, G.M. Feeding and management practices affect quality improvement in wild-caught edible crab (*Cancer pagurus*). *Aquaculture* **2007**, *269*, 328–338. [CrossRef]
39. Flores, A.; Wiff, R.; Ganias, K.; Marshall, C.T. Accuracy of gonadosomatic index in maturity classification and estimation of maturity ogive. *Fish. Res.* **2019**, *210*, 50–62. [CrossRef]
40. Barrento, S.; Marques, A.; Teixeira, B.; Anacleto, P.; Vaz-Pires, P.; Nunes, M.L. Effect of Season on the Chemical Composition and Nutritional Quality of the Edible Crab *Cancer pagurus*. *J. Agric. Food Chem.* **2009**, *57*, 10814–10824. [CrossRef] [PubMed]

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Catch per Unit Effort of Decapod Species, *C. pagurus* and *H. gammarus*, from a Voluntary Marine Reserve

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Abstract: *C. pagurus* and *H. gammarus* are deemed to be declining in abundance in the Berwickshire Marine Reserve from personal communications with local inshore fishers. Fisheries data in the form of catch per unit effort (CPUE) were collected for these two commercially important decapods. Other explanatory variables from fishing activity such as the creel and bait type used, the soak time of the fishing gear, and deployment depth were recorded to provide as much detail as possible to describe the effort applied to catch these decapod species. In this study, CPUE was higher for *H. gammarus* and *C. pagurus* outside the Berwickshire Marine Reserve. General additive models (GAMs) were used to describe the effects of the explanatory variables and showed that soak time (days) and depth (m) significantly affected CPUE for *C. pagurus*, not *H. gammarus*. Sea temperature (°C) showed a negative correlation with the CPUE of both *H. gammarus* and *C. pagurus*; however, a positive correlation was found with the number of *C. pagurus* caught. The data collected in this study provide a foundation in understanding the current abundance of *C. pagurus* and *H. gammarus* in a voluntary marine reserve on the east coast of Scotland, which can be used to inform future changes in fisheries management in Berwickshire.

Keywords: catch per unit effort; *Cancer pagurus*; *Homarus gammarus*; fisheries

Key Contribution: The abundance of *C. pagurus* and *H. gammarus*, described in the form of catch per unit effort, highlights a potential overexploitation of the commercially important species inside the Berwickshire Marine Reserve.

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1. Introduction

Pressure is applied to shellfish fisheries, such as *C. pagurus* and *H. gammarus*, to compensate for recovering finfish fisheries for food security [1,2]. However, it is suggested that *C. pagurus* and *H. gammarus* are deemed "fish to avoid" by the Marine Conservation Society [3]. With contradicting statements, there is a clear need for as much information as possible to fully understand the sustainability status of commercially important species in the Berwickshire region. Personal observations highlighted that many inshore fishers suggest a decline of both *H. gammarus* and *C. pagurus* in the region, which these fishers fully depend on for financial support. *C. pagurus* stocks are currently misunderstood regarding their current level of exploitation [4]. In the region of Berwickshire, based on statistics from Eyemouth Harbour (55.871697, −2.087245), there are around 62 creel fishing vessels under 10 m in length and 3 vessels over 10 m typically fishing for *C. pagurus* and *H. gammarus* [5]. These vessels are below 12 m in length and are regarded as part of the inshore fishing fleet [6,7]. In the 10.3 km^2 Berwickshire Marine Reserve, fishers contribute to 91–97% of the annual value of landings in the U.K., of which in 2019 this contribution was 19% [5,8]. In 2020, the *C. pagurus* fishery for Berwickshire (Eyemouth Harbour) was valued at GBP 567,000, and the *H. gammarus* fishery was valued at GBP 2,565,000 [5]. Although generating valuable annual income to the U.K. economy and to the local economy of Berwickshire, it is believed that U.K.-wide *C. pagurus* and *H. gammarus* fisheries are overexploited or close to overexploited [2].

Catch per unit effort (CPUE) is a common tool for monitoring and reporting fish populations through analysis of commercial landings to provide an abundance index [2,9,10]. If the resultant CPUE is declining, this may indicate a fishery that cannot support the level of fishing it experiences; likewise, an increasing CPUE can indicate the fished stock is recovering and potential increases in fishing activity may be applied with appropriate management [11]. In addition to landings data, morphometric data (wet weight (g) and carapace length/width (mm)), can be added to understand the growth and health of the fished stock [12]. With all forms of fishing, there are variables that can influence the fishers' catch. Pots, or creels, are a form of fishing using various bait types. They can be fitted with panels for undersized or unwanted bycatch to escape and can vary structurally with the inclusion or exclusion of one or more "eyes" (the entrance to a fishing creel). Inter- and intra-specific interactions between individuals within the vicinity of the fishing creel can also influence the individuals that enter and leave a creel [13]. In addition, the season [14], soak time [15,16], and environmental changes can potentially lead to a misinformed calculation of CPUE, abundance, and size distribution for a locale [15,17,18]. To understand such a complex system, these variables should be considered in fisheries assessments, although they currently are not; therefore, interpreting CPUE data fully should be undertaken with caution. Furthermore, collating specific information from individual fishing vessels such as the location of creel hauling, the number of creels per fleet, and soak time for each fleet can provide a full description of the CPUE, particularly for mixed trap fisheries [2]. CPUE is generally standardised using statistical methods such as general linear models (GLMs) and general additive models (GAMs), which allow for descriptions of CPUE with greater detail by considering such variables; however, this still does not provide a full descriptor for abundance in real terms as it cannot correlate well with the statistical output of such tests [19,20].

Marine reserves and other protected marine designations are becoming more apparently used to mitigate the declining biodiversity currently observed worldwide [21–23]. Most of these designations are backed by legislation; however, in the case of the Berwickshire Marine Reserve, it is a voluntary designation. The Berwickshire Marine Reserve is one of three other statutory reserves on the Berwickshire coastline; of these, one is a special area of conservation, the Berwickshire and North Northumberland SAC, and the other prevents the use of mobile fishing gear, the Static Gear Reserve (Figure 1). Therefore, static gear fishing is the primary form of fishing within and outside the Berwickshire Marine Reserve. At present, the only management protocol for this fishery is the current minimum landing size for both *C. pagurus* and *H. gammarus*, which is 150 mm and 90 mm, respectively [24]. The number of creels associated with each inshore fisher is not limited, and therefore, there is an increasing interest in creel limitations, which have started with pilot studies on the west coast of Scotland [25]. Currently, there is no scope for such a limitation in Berwickshire, and anecdotal data from local inshore fishers suggest some vessels are deploying over 1000 creels, with one extreme case of 10,000 creel deployments. Inshore fishing vessels are not required at present to utilise vessel tracking technology, and therefore, monitoring of these vessels' fishing activity regionally is not common practice either. However, this is soon to change with the current inshore modernisation programme detailed in the "Bute House Agreement" and "fisheries management strategy 2020 to 2030 delivery plan" which aims to enhance monitoring activities of the inshore fishing fleet using onboard vessel technology [26] as this sector is poorly reported and understood. The data collected from these monitoring systems will allow areas of fishing activity to be monitored, which can aid the understanding of fishing effort in a more regional context. Without current context and understanding of the CPUE in Berwickshire, the means of assessing the effects of new management or legislation are limited.

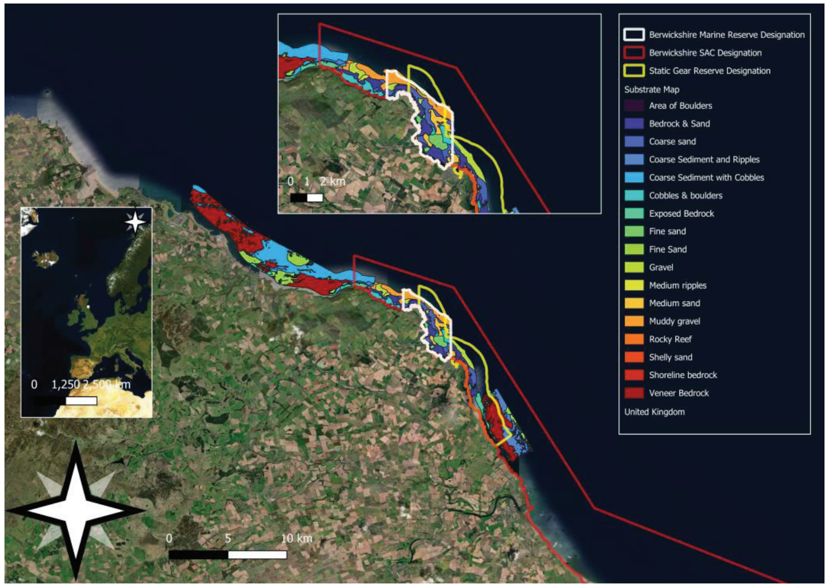

Figure 1. Habitat map of the Berwickshire coastline (55.912714, −2.109902) that includes the mixing of substrate types in and outside the Berwickshire Marine Reserve designation. The habitat map was provided by Blue Marine Foundation.

This study aims to provide information for a data-limited fishery in a voluntary marine reserve to understand the local *C. pagurus* and *H. gammarus* abundance in the Berwickshire Marine Reserve. We incorporate local inshore fishers that conducted independent onboard assessments from 2018 to 2019 to collate information on CPUE for these species. Included in the assessments are bait type, creel type, and environmental variables, such as sea temperature (°C), dissolved oxygen (%), and salinity (ppt) which cover many of the factors that could influence CPUE.

2. Materials and Methods

2.1. Study Area

The Berwickshire Marine Reserve (55.912714, −2.109902) covers an open sea area of 10.3 km^2 (Figure 1) which extends out from the Berwickshire coastline to a 50 m depth contour. Within this designation, the habitat is formed from rocky outcrops interspersed with patches of sand (Figure 1). The BMR is situated within the Berwickshire and North Northumberland Special Area of Conservation (SAC). The SAC covers an area of 652 km^2, from Alnmouth in the south to Fast Castle Head in the north, which includes the St. Abbs and Eyemouth Static Gear Reserve which covers 26 km^2 and extends one nautical mile offshore from St. Abbs Head in the north to the Scotland–England Border in the south. Onboard independent surveys took place within the Berwickshire Marine Reserve, Southeast Scotland, U.K. (55.912714, −2.109902), and outside this designation (Figure 2). In the local area, the fishing season for both species is typically all year round; however, the intensity of fishing activity is greater in the summer months for *H. gammarus* and in the winter and spring months for *C. pagurus*, which can be indicated by the fluctuating annual market value of these decapods.

Figure 2. Creel fleet locations from all CPUE surveys between 2018 and 2019. Creel fleets are separated by colour representing each fishing vessel used. The white outline highlights the Berwickshire Marine Reserve designation (55.912714, −2.109902). Orange and yellow dots were categorised as inside the Berwickshire Marine Reserve designation, and red, green, and blue are located along the Berwickshire coast and categorised as outside the marine reserve.

2.2. Onboard Surveys

In the period between 2018 and 2019, catch per unit effort (CPUE) surveys were conducted using 5 local inshore fishing vessels of sizes 12 m or less. The local fishing vessels used were berthed in either St. Abbs Harbour (55.899190, −2.129224) or Eyemouth Harbour (55.871697, −2.087245). Surveys were conducted throughout the primary fishing season, with the number of surveys decreasing towards the winter months due to lower fishing activity in response to adverse weather. Surveys generally occurred between the times of 0600 h–1500 h, with each survey varying in duration from 1 to 8 h. Multiple creels are fished as fleets. The number of creels deployed in each surveyed fleet reached a maximum number of 45, with some surveys including single pots which are known as single enders in the local area. Fleets were deployed in a random method, as fishers deployed fleets as they would during normal fishing activity. The creels used by the local inshore fishers varied from single hard-eye to double soft-eye parlours (Figure 3). The variation between the creel types used is dependent on the entrance, known as the eye, and the number of eyes per creel. Eyes can be either made from netting, termed soft eyes, or from a hard plastic 6-inch ring termed a hard eye. Typically, the rest of the creel structure is built from a 38-inch frame of either plastic or metal and laced with rope made from a nylon material (Figure 3). Each fleet of creels was deployed in a depth range between 5.2 m and 32 m which varied depending on the fishing vessel used and the month of the year. The length of time creels were deployed, known as soak time, was recorded in days and ranged from 1 to 7 days. Creels used in this study were typically not fitted with escape panels as this is common for the area. All vessel names and locations of creel retrievals were recorded using their onboard GPS or by handheld GPS which all onboard surveyors carried. The recording of all CPUE data was conducted by two members of staff from St. Abbs Marine Station and two staff rangers from the Berwickshire Marine Reserve (BMR).

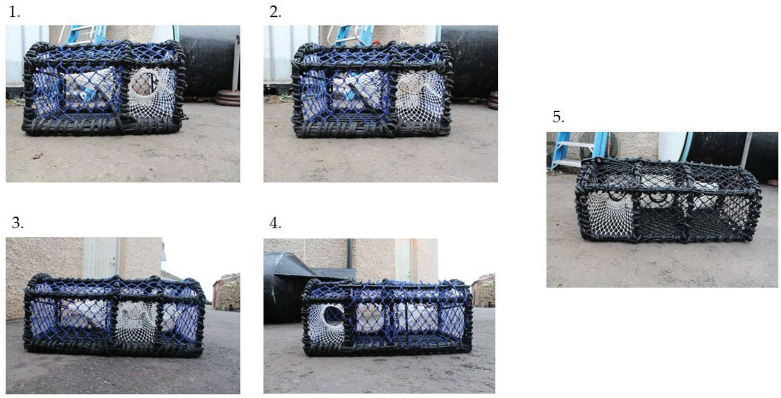

Figure 3. Creel types used from all fishing vessels during the sampling period. No escape panels were used on any of the creels sampled over the survey period. The creels shown in the figure were not used during the study. These are used to present a visual representation of the creels utilised by the local inshore fishers. Image 1 shows a single hard-eye parlour and image 2 is a single soft-eye parlour. Image 3 shows a double soft-eye parlour and image 4 is a double hard-eye parlour. Lastly, image 5 shows a prawn parlour which is smaller in size and is fitted with two entrances.

2.3. CPUE

Catch per unit effort (CPUE) (kg per day^{-1}) was calculated for each onboard survey. For data collected in 2019, landing per unit effort (LPUE) (kg per day^{-1}) and net retrieval per unit effort (NRPUE) (kg per day^{-1}) were also measured. LPUE is the number of individuals of landing size (>85 mm *H. gammarus*; >140 mm *C. pagurus*) or above based on the number of creels hauled, and in contrast, the NRPUE is the number of individuals below the landing size.

$$\text{CPUE} = \frac{\text{Total number of target species per fleet}}{\text{Number of creels per fleet}}$$

$$\text{LPUE} = \frac{\text{Total number of landing sized individuals per fleet}}{\text{Number of creels per fleet}}$$

$$\text{NRPUE} = \frac{\text{Total number of undersized individuals per fleet}}{\text{Number of creels per fleet}}$$

All bycatch was recorded using the common species name and the number of individuals present in each hauled creel. As the number of *Necora puber*, which are deemed bycatch by the inshore fishers, caught was so high, they were included in the CPUE assessments as they can be collected and sold by the local inshore fishers.

2.4. Environmental Data

Environmental data such as sea surface temperature (°C), dissolved oxygen (mg/L), salinity (ppt), and lunar phase (%) were recorded on each day an onboard survey was conducted. These data, except lunar phase, were collected using a YSI ProSolo digital water quality meter probe deployed at sea before each fleet haul. Data were rounded to two decimal places for analysis. Lunar phase data were collected on each onboard survey day from a meteorological website (Willyweather.co.uk accessed on 1 June 2018).

2.5. Statistical Analyses

All statistical analysis was conducted using R Studio (Version 1.1.463–© 2009–2018 RStudio, Inc., Boston, MA, USA) [27]. Data were inspected for normality of distribution and equal variance using the Shapiro–Wilks test and Bartlett's test. Histogram plots were used for visual inspection of normality distribution. Unpaired two-sample t-tests were used to compare the number of individuals caught inside and outside the Berwickshire Marine Reserve designation. To find any differences in the number of individuals caught by year and soak time, Kruskal–Wallis tests were conducted. Pairwise Wilcox tests were used to further the analysis of the Kruskal–Wallis tests to include month as a factor. Kendall rank tau correlations were conducted to consider the effects of environmental variables such as sea surface temperature (°C), salinity (ppt), and dissolved oxygen (mg/L) on the number of target species caught and CPUE. Tau values of above 0 were indicative of positive relationships, and values above 1 were regarded as highly positive. Tau values of below 0 were indicative of negative relationships, with values below -1 regarded as highly negative values.

General additive models (GAMs) were used to model the relationships of CPUE for each target species in relation to the explanatory variables, as shown by the following:

$$\text{CPUE} \sim \text{year} + \text{s(depth)} + \text{s(soak time)} + \text{boundary} + \text{bait} + \text{creel type}$$

The complexity of the models was reduced by comparing the REML scores (restricted maximum likelihood) and R^2 values. Models with a low REML score when compared to the other tests were favoured. The models were fitted with a quasi-Poisson function due to overdispersion in the initial models. Soak time (days) and depth (m) were selected as isotropic smooths (s) based on the models used in [20]. Chi-square tests were then used to assign which variables significantly influenced the CPUE of the target species. Significance was assumed when the *p*-value was <0.05 for all statistical tests.

3. Results

3.1. Catch Composition

From both sampling years collectively, the total number of *H. gammarus* individuals reached $n = 3013$, the total number of *C. pagurus* individuals reached $n = 2117$, and the total number of *N. puber* individuals reached $n = 3388$. These numbers were collected across a total of 23 surveys and 1897 individual creel pots (Table 1). In 2019, the number of undersized individuals (<minimum landing size (<MLS)) was 834 for *H. gammarus* and 427 for *C. pagurus*. The soak time to catch the greatest number of *H. gammarus* (>MLS) was 7 days, although this was deemed insignificant ($p = 0.07$). However, for those *H. gammarus* <MLS, significance was found for all soak times above 2 days ($p \leq 0.05$). A soak time of 5 days showed significance for the highest number of *C. pagurus* (<MLS) caught ($p < 0.05$). Contrastingly, a soak time of 2 and 3 days proved to be significant compared to 7 days ($p = 0.03$) when related to the number of *N. puber* caught.

Table 1. Total count of the boats, surveys, and creels used in this project. Data are separated by year and show the total count for each group. Total number of *H. gammarus*, *C. pagurus*, and *N. puber* caught over the surveys is shown.

Year	No. of Boats	No. of Surveys	No. of Creels	No. of *H. gammarus*	No. of *C. pagurus*	No. of *N. puber*
2018	1	8	472	389	567	550
2019	5	15	1425	2624 (834 < MLS)	1550 (427 < MLS)	2838

Note: *N. puber* are not distinguished by a minimum landing size, and therefore, values are not separated similar to *H. gammarus* and *C. pagurus* in the table. MLS is the abbreviation for minimum landing size (minimum landing size 150 mm for *C. pagurus* and 80 mm for *H. gammarus*).

The number of *H. gammarus* (chi-square = 7.36, $p = 0.02$) and *C. pagurus* (chi-square = 32.37, $p \leq 0.05$) decreased from 2018 to 2019, whereas the number of *N. puber* (chi-square = 83.60, $p \leq 0.05$) increased. The number of *H. gammarus* increased between July (2018: 1.43 $\pm$ 0.07, 2019: 1.26 $\pm$ 0.05) and August (2018: 1.67 $\pm$ 0.06, 2019: 1.35 $\pm$ 0.09) ($p = 0.01$). For *C. pagurus*, in 2018, numbers decreased from July (2.28 $\pm$ 0.10) to August (1.66 $\pm$ 0.10), whereas in 2019, there was not a large discrepancy (July: 1.10 $\pm$ 0.05, August: 1.18 $\pm$ 0.13) ($p \leq 0.05$).

For *N. puber*, the number of individuals caught in August differed from those in July and September ($p \leq 0.05$).

3.2. CPUE

CPUE data from 2018 for both *C. pagurus* and *H. gammarus* was highest in prawn parlour creel variations (n = 2.92 and 1.85). The lowest values were recorded in double-eye creels for *C. pagurus* (n = 0.94) and single-eye creels for *H. gammarus* (n = 0.57) (Table 2). The CPUE of *N. puber* was found to be highest in single-eye creels (n = 2.26) and lowest in double-eye variations (n = 0.83). In 2019, CPUE for *H. gammarus* was highest in hard-and-soft-eye creels and lowest in prawn parlours (Table 3). For *C. pagurus*, CPUE was highest in parlours and lowest in prawn parlours (Table 3). For *N. puber*, CPUE was highest in prawn parlours and lowest in hard- and soft-eye creels, respectively (Table 2). In data from 2019, NRPUE was greater than LPUE for all target species. NRPUE was highest for *H. gammarus* (n = 1.29), whereas LPUE was lowest for *C. pagurus* (n = 0.31).

Table 2. CPUE for all target species in 2018–2019 based on pot types sampled.

Year	Creel Type	No. of Pots	No. of *H. gammarus* ≥MLS/≤MLS	No. of *C. pagurus* ≥MLS/≤MLS	CPUE (L)	CPUE (C)	CPUE (V)
2018	Double-Eye	311	293	291	0.942	0.942	0.836
	Single-Eye	146	70	235	0.570	1.520	1.650
	Prawn Parlour	14	26	41	1.857	2.928	1.571
2019	Double-Eye Parlour	109	101/148	36/93	2.284	1.183	1.12
	Double Soft-Eye Parlour	110	66/73	35/21	1.263	0.509	2.8
	Hard- and Soft-Eye	8	7/16	0/7	2.875	0.875	0.25
	Hard-Eye Parlour	329	143/291	57/123	1.32	0.55	2.29
	Parlour	624	335/988	301/749	2.120	1.682	1.639
	Prawn Parlour	44	21/21	1/5	0.954	0.136	3.659
	Soft-Eye Parlour	144	86/302	25/73	2.69	0.68	2.36

Note: CPUE values for *H. gammarus* (L), *C. pagurus* (C), and *N. puber* (V) are presented for each pot type sampled. Values are calculated by the number of individuals caught divided by the number of pots of the specific creel type. The values shown for total number of *H. gammarus* and *C. pagurus* are separated by landing size (≥MLS)/undersized (≤MLS). Numbers of *H. gammarus* and *C. pagurus* in 2018 were grouped and not separated by landing size, as represented in the table.

Table 3. CPUE (2018–19) (mean ± standard error mean) of the target species collected from all creels hauled by the respective fishing vessel.

Year	Vessel	BMR Designation	No. of Pots	CPUE (L)	CPUE (C)	CPUE (V)
2018	Vessel 1	In	468	0.82 ± 0.02	1.12 ± 0.04	1.17 ± 0.04
2019	Vessel 1	In	453	1.00 ± 0.02	0.37 ± 0.01	2.86 ± 0.06
	Vessel 2	In	256	1.28 ± 0.05	1.33 ± 0.05	1.53 ± 0.05
	Vessel 3	Out	47	3.72 ± 0.21	0.17 ± 0.03	1.34 ± 0.13
	Vessel 4	Out	370	2.75 ± 0.04	1.31 ± 0.05	1.47 ± 0.05
	Vessel 5	Out	299	2.16 ± 0.04	1.80 ± 0.07	1.80 ± 0.06

Note: Vessels are also separated by their fishing area inside or outside the Berwickshire Marine Reserve designation.

Regarding CPUE of the target species inside and outside the Berwickshire Marine Reserve, for *H. gammarus*, it was greater outside (2.57 ± 0.03) than inside (1.10 ± 0.02) the designation (t = 41.65, $p \leq 0.05$). This was also the case for *C. pagurus* (outside: 1.44 ± 0.04; inside: 0.72 ± 0.02) (t = 11.30, $p \leq 0.05$). Only the CPUE for *N. puber* was greater inside (2.38 ± 0.04) than outside (1.60 ± 0.04). Regarding the number of these target species, both *H. gammarus* (≥MLS: 1.51 ± 0.04; ≤MLS: 2.39 ± 0.05) and *C. pagurus* (≥MLS: 1.62 ± 0.07;

≤MLS: 2.91 ± 0.18) were higher outside the designation. The number of *N. puber* caught was greater inside (3.59 ± 0.11) the designation than outside (3.30 ± 0.12) (t = −5.13, $p \leq 0.05$).

In the analysis of GAM models of CPUE of both *H. gammarus* and *C. pagurus*, it was found that the simpler model (CPUE ~ year + s(depth) + s(soak time) + boundary) was deemed suitable based on the chi-square test results and REML scores ($p \leq 0.05$) (Table 4). For *N. puber*, the more complex model was deemed more suitable in describing the changes in CPUE (CPUE ~ year + s(depth) + s(soak time) + boundary + bait + pot type) (df = 15.95, $p \leq 0.05$). The outputs from all models can be seen in Table 4. Considering each model, soak time (days) was significant for CPUE for all target species ($p \leq 0.05$) (Figures 4–6). Depth (m) also showed a significant effect on the model and CPUE for all models except for *H. gammarus* (F = 1.87, $p = 0.19$) (Figure 5).

Table 4. General additive model outputs describing the relationship of CPUE of all target species with depth, soak time, bait, creel type, and area in relation to the Berwickshire Marine Reserve designation.

Species	Model	REML	R^2	Deviance	Intercept: *t*-Value	Intercept: *p*-Value
H. gammarus	Model A	−1381.5	0.63	63.9%	−0.30	0.76
	Model B	−1575.7	0.72	71.7%	−6.26	<0.05
	Model C	−879.7	0.52	41.7%	−8.76	<0.05
C. pagurus	Model A	−1141.1	0.63	62.6%	16.27	<0.05
	Model B	−838.2	0.46	46.8%	1.66	0.09
	Model C	−580.83	0.33	31.2%	7.27	<0.05
N. puber	Model A	−547.58	0.18	15.8%	−18.12	<0.05
	Model B	−686.55	0.25	26.1%	−10.69	<0.05
	Model C	−895.07	0.42	42.7%	−0.33	0.73

Note: Results include all data from all 5 fishing vessels across both survey years (2018–19). Significance was set at $p < 0.05$. The models used are as follows: model A: CPUE ~ year + s(depth) + s(soak time) + boundary + bait; model B: CPUE ~ year + s(depth) + s(soak time) + boundary + bait + creel type; model C: CPUE ~ year + s(depth) + s(soak time) + boundary. Best-fitting models were selected based on the REML score (the lower the score, the better the description of CPUE) and R^2 value (the greater the value, the better the fit to the model). Significance was set at $p < 0.05$. Chi-square tests were used to compare and test which models were best for the CPUE data.

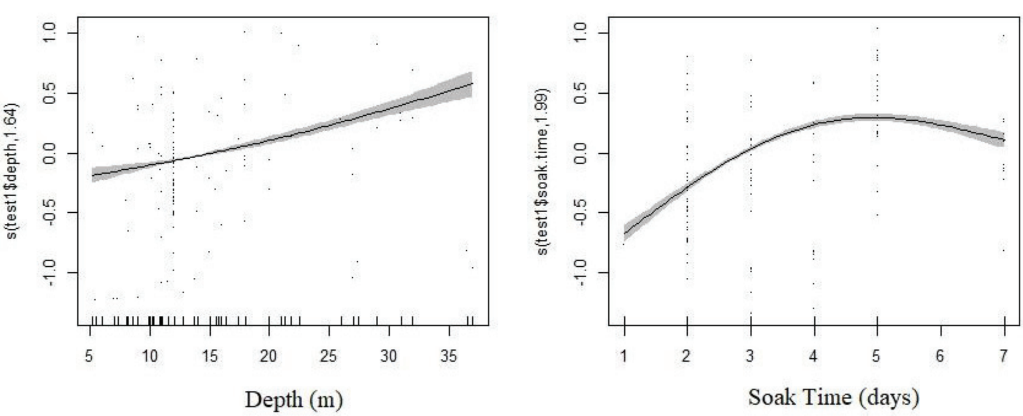

Figure 4. Effects of smoothing parameters depth and soak time on the CPUE of *C. pagurus* following GAM model analysis of data from 2018 to 19. Graphs are produced from the best-fitting model (CPUE ~ year + s(depth) + s(soak time) + boundary). Smoothing parameters were assigned basis functions in the model as s(parameter, k = 3); this set the maximum possible degrees of freedom for the smoothing parameter as 3.

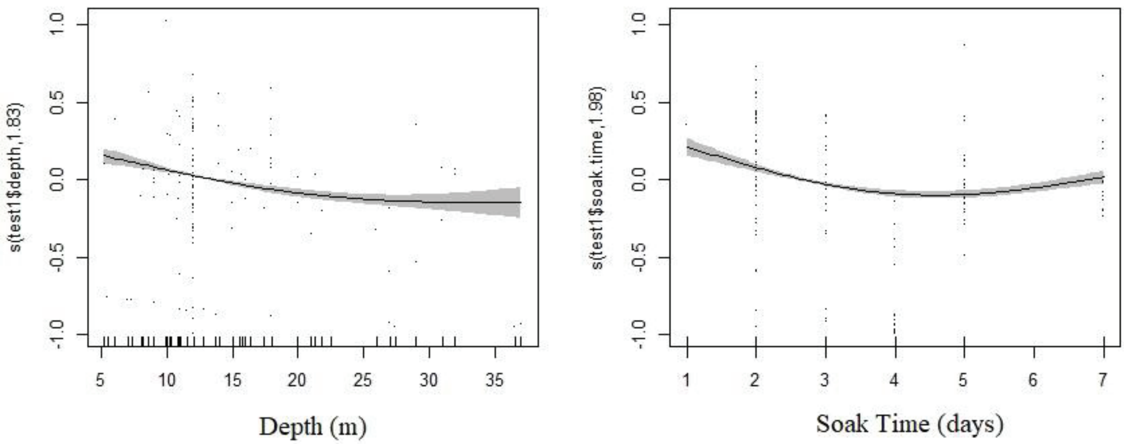

Figure 5. Effects of smoothing parameters depth and soak time on the CPUE of *H. gammarus* following GAM model analysis of data from 2018 to 19. Graphs are produced from the best-fitting model (CPUE ~ year + s(depth) + s(soak time) + boundary). Smoothing parameters were assigned basis functions in the model as s(parameter, k = 3); this set the maximum possible degrees of freedom for the smoothing parameter as 3.

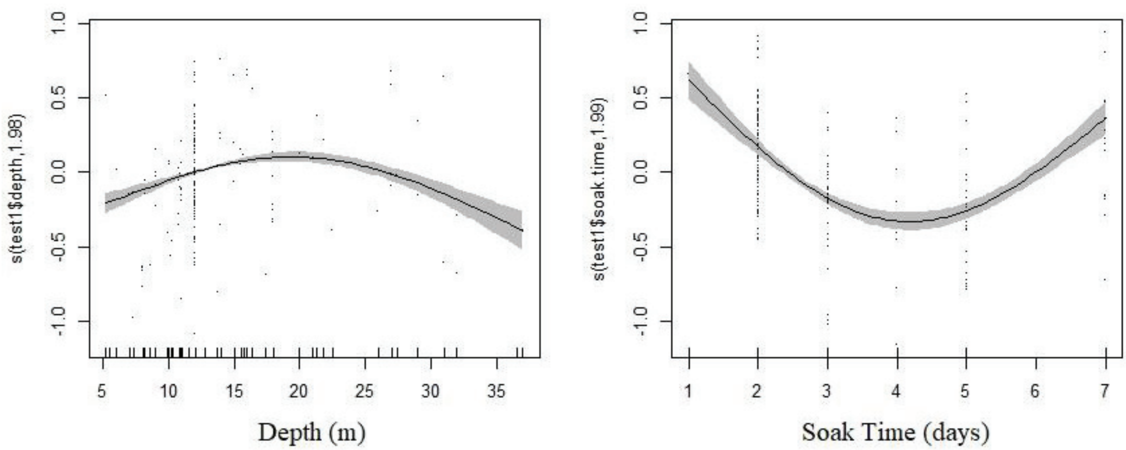

Figure 6. Effects of smoothing parameters depth and soak time on CPUE of *N. puber* following GAM model analysis of data from 2018 to 19. Graphs are produced from the best-fitting model (CPUE ~ year + s(depth) + s(soak time) + boundary + bait + pot type). Smoothing parameters were assigned basis functions in the model as s(parameter, k = 3); this set the maximum possible degrees of freedom for the smoothing parameter as 3.

3.3. Environmental Data

Between the sampling years of 2018 and 2019, August was the warmest month (sea surface temperature) (2018: 15.38 ± 0.26 °C; 2019: 15.08 ± 0.21 °C). The coldest month with respect to sea surface temperature was March in 2018 (4.77 ± 0.29 °C) and February in 2019 (6.15 ± 0.09 °C). April for both years had the lowest recorded salinity (2018: 33.57 ± 0.06 ppt; 2019: 33.68 ± 0.21 ppt), with September recording the highest in 2018 (34.44 ± 0.04 ppt) and October recording the highest in 2019 (34.39 ± 0.03 ppt). A full breakdown of the environmental variables recorded can be seen in Table 5.

Table 5. Environmental variables (sea temperature (°C), salinity (ppt), and dissolved oxygen (mg/L)) recorded by month for each sampling year (2018–2019). Data shown as mean ± sem.

	2018		**2019**		
Month	**Sea Surface Temperature (°C)**	**Salinity (ppt)**	**Sea Surface Temperature (°C)**	**Salinity (ppt)**	**Dissolved Oxygen (mg/L)**
June	11.80 ± 0.18	34.13 ± 0.02	12.06 ± 0.24	33.91 ± 0.03	9.15 ± 0.06
July	14.80 ± 0.52	34.16 ± 0.05	14.33 ± 0.25	33.88 ± 0.07	8.38 ± 0.06
August	15.38 ± 0.26	33.90 ± 0.30	15.08 ± 0.21	33.86 ± 0.06	8.01 ± 0.25
September	12.75 ± 0.28	34.44 ± 0.04	12.61 ± 0.08	34.22 ± 0.09	8.84 ± 0.17
October	10.88 ± 0.20	34.41 ± 0.04	10.77 ± 0.15	34.39 ± 0.03	8.89 ± 0.09
November	9.99 ± 0.17	34.24 ± 0.04	9.24 ± 0.16	33.78 ± 0.21	9.33 ± 0.07
December	8.40 ± 0.00	34.31 ± 0.00	7.80 ± 0.15	34.00 ± 0.24	9.43 ± 0.05

H. gammarus of sizes above the MLS were not positively or negatively related to environmental factors significantly (Table 6). However, those below the MLS were positively correlated with dissolved oxygen (tau = 0.18, $p \leq 0.05$), salinity (tau = 0.15, $p \leq 0.05$), and lunar phase (tau = 0.11, $p \leq 0.05$) and negatively correlated with sea surface temperature (tau = −0.18, $p \leq 0.05$) (Table 6). *C. pagurus* individuals of sizes below the MLS followed the same trend as the undersized *H. gammarus*; however, there was no significant result in correlation with salinity (tau = 0.05, $p = 0.15$). Those *C. pagurus* individuals above the MLS were positively correlated with sea surface temperature (tau = 0.14, $p \leq 0.05$) and negatively correlated with dissolved oxygen (tau = −0.09, $p \leq 0.05$) and lunar phase (tau = −0.10, $p \leq 0.05$). CPUE of both *C. pagurus* and *H. gammarus* was tested with lunar phase and sea surface temperature (Figure 7), and it was found that there was a weak negative relationship for both species regarding sea surface temperature (tau ≤ −0.5, $p \leq 0.05$) and a weak positive relationship for lunar phase (tau ≥ 0.5, $p \leq 0.05$) (Table 6).

Table 6. Kendall rank correlation outputs (*p*-value and tau value) for abundance counts and CPUE related to environmental variables for *C. pagurus* and *H. gammarus*.

Species		**Individuals ≥ MLS**	**Individuals ≤ MLS**	**CPUE**
H. gammarus	Sea Temperature (°C)	$p = 0.37$ tau = 0.02	$p \leq 0.05$ tau = −0.18	$p \leq 0.05$ tau = −0.61
	Salinity (ppt)	$p = 0.27$ tau = 0.03	$p \leq 0.05$ tau = 0.15	
	DO (mg/L)	$p = 0.05$ tau = −0.05	$p \leq 0.05$ tau = 0.18	
	Lunar Phase (%)	$p = 0.70$ tau = −0.01	$p \leq 0.05$ tau = 0.11	$p \leq 0.05$ tau = 0.46
C. pagurus	Sea Temperature (°C)	$p \leq 0.05$ tau = 0.14	$p = 0.00$ tau = −0.10	$p \leq 0.05$ tau = −0.21
	Salinity (ppt)	$p = 0.84$ tau = −0.00	$p = 0.15$ tau = 0.05	
	DO (mg/L)	$p = 0.00$ tau = −0.09	$p \leq 0.05$ tau = 0.17	
	Lunar Phase (%)	$p = 0.00$ tau = −0.10	$p \leq 0.05$ tau = 0.21	$p \leq 0.05$ tau = 0.34

Note: Significance was set at $p < 0.05$. Negative tau values correspond to negative relationships and positive tau values correspond to positive relationships. The greater the tau value ≥ 1, the greater the positive relationship; and with tau values ≤ −1, the greater the negative relationship. DO denotes dissolved oxygen (mg/L).

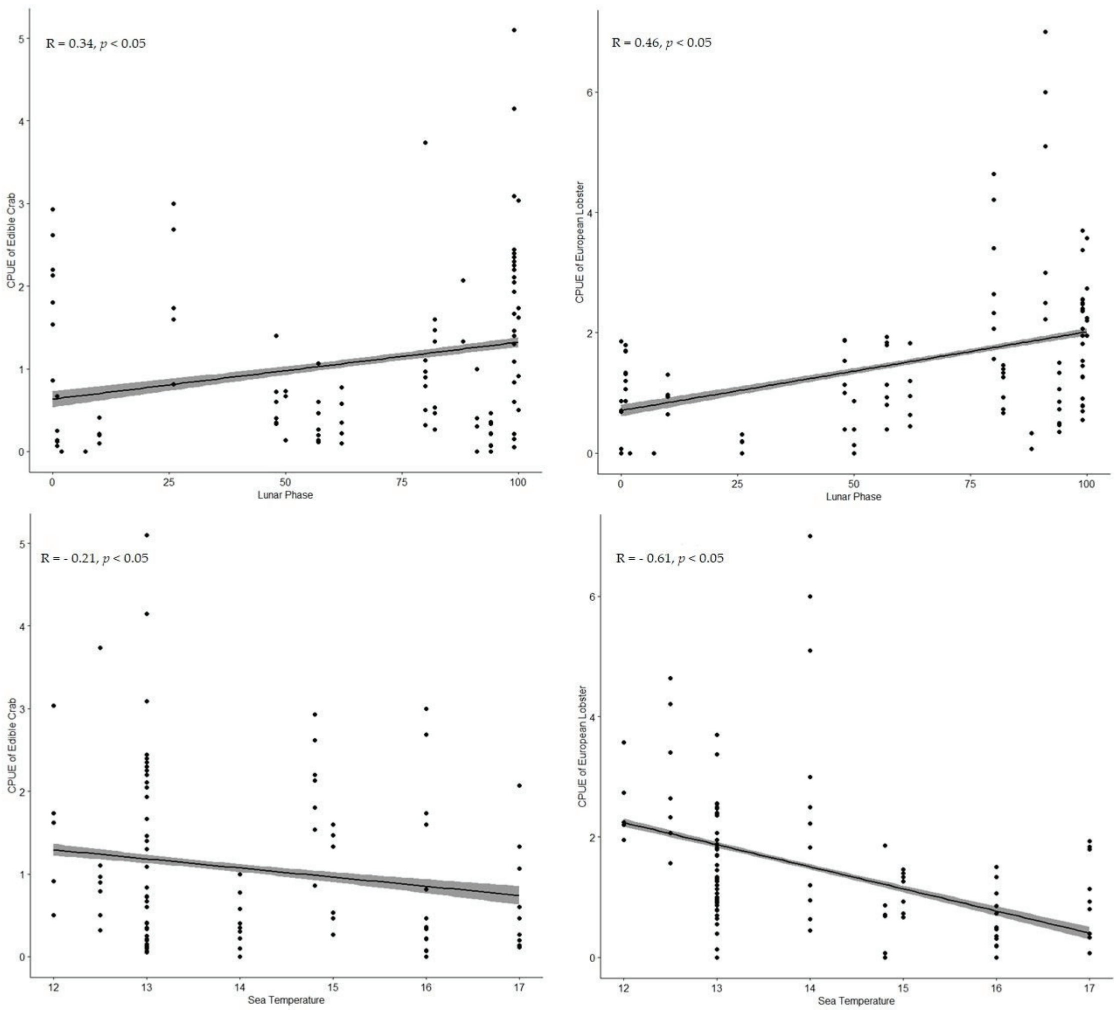

Figure 7. Kendall rank correlation of lunar phase (%) and sea surface temperature (°C) with respect to CPUE of *C. pagurus* and *H. gammarus*. This included undersized and landing-sized individuals. Black dots represent the recorded CPUE for the target species from each survey conducted over the study. A positive correlation for lunar phase (%) is highlighted by the trendline with an R value of 0.34 ($p \leq 0.05$) for *C. pagurus* and 0.46 ($p \leq 0.05$) for *H. gammarus*. Negative correlations for sea temperature are highlighted by the trendline with an R value of −0.21 ($p \leq 0.05$) for *C. pagurus* and −0.61 ($p \leq 0.05$) for *H. gammarus*. Significance was set at $p < 0.05$.

3.4. Bait

The type of bait used by the fishers can be seen in Figure 8, with a greater proportion of mackerel (*Scomber scombrus*) (42.07%) and herring (*Clupea harengus*) (34.35%) used across all fishing vessels. Bait was included in the GAM models, in which mackerel was significantly related to the number of *H. gammarus* ($p = 0.01$) and *C. pagurus* ($p = 0.01$) caught. Herring as a bait source was significant for the number of *C. pagurus* caught ($p \leq 0.05$). Contrastingly, scad (*Trachurus trachurus*) showed significance for the number of *N. puber* caught ($p \leq 0.05$).

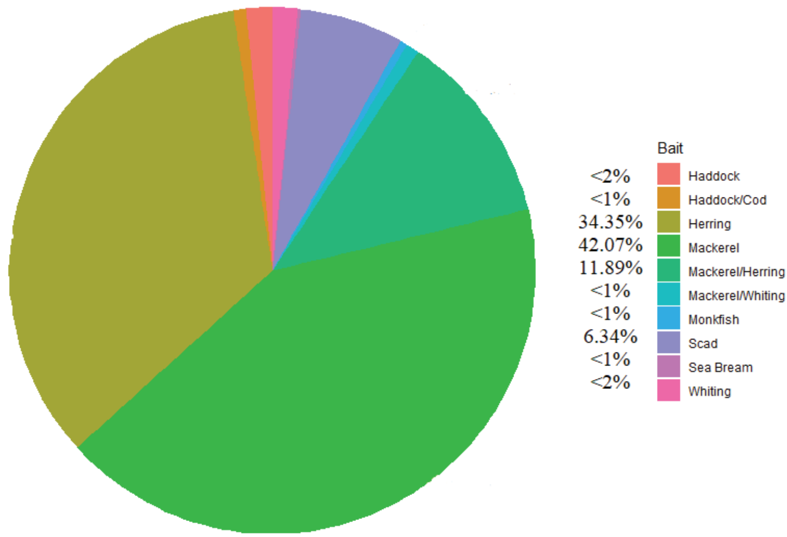

Figure 8. Pie chart of the bait type used across 2018 and 2019. Data are separated by bait type used across all fishing vessels. The data shown are the percentage of creels using the bait types highlighted in the figure.

4. Discussion

4.1. Catch Composition

In the local area, it is a common theory with local fishers that the stocks for both *H. gammarus* and *C. pagurus* are declining (pers. obvs.). Previous studies have implemented the use of logbooks, used by commercial fishers, to collate the information in situ; however, the robustness of this assessment is determined by the number of fishers participating [15,20]. In this study, a low number of fishing vessels (n = 5) were observed independently using onboard observers without the use of fisher logbooks. Logbooks are used to record the annual landings of fishers' catch every time they fish at sea. In this study we did not observe every fishing trip; therefore, it may be assumed that some detail and nuance have been lost from the results, which may have been provided if logbooks were used. It has been addressed that due to the limited number of vessels fishing used in this study, there is potential that the catch could be more uniform than would be representative of the natural stock, a positive intra-cluster correlation [15,28]. However, by utilising fishers with different fishing grounds and fishing fleets of randomised locations, it is perceived that this positive intra-cluster correlation was reduced. Many factors could influence the *C. pagurus* populations regionally, such as habitat variations, mating behaviour, and competition for food and shelter [20]. The influence is based on the sex ratio of caught individuals, which differs seasonally and spatially with *C. pagurus* migrating in the Autumn after emigrating back to inshore grounds to mate and moult [17,20,29]. The habitats outside and inside the marine reserve designation are predominantly rocky outcrops with interspersed sandy patches with the latter increasing in presence outside the marine reserve. Such habitats benefit the female *C. pagurus* who reside in sandy patches to avoid strong currents and to incubate eggs post-mating [20,30,31] (Figure 1). The rocky outcrops may have led to lower numbers of *H. gammarus* and *C. pagurus* caught due to the availability of natural shelter outweighing exploratory behaviour around creels. *H. gammarus* tend to shelter in the late spring as they moult and harden their shell before re-emerging in the late summer [32,33], which coincides with the increase in CPUE in August and July in 2019 shown in this study. The number of *C. pagurus* caught has been shown to be related to sediment type following GAM models in [4] which showed that gravel substrate proved to have higher catch rates than softer sediment. This was not included in our models as the locations of creel pots could not be "ground-truthed" to specific substrate types as the habitat map described in Figure 1 was provided after sampling and analysis. The study presented in [4] also used dredges and trawls, a form of mobile fishing, which is not comparable to static gear fishing which involves the need for food-seeking behaviour using odour cues [34,35].

4.2. CPUE

Although independent observations typically are low in replicates, they are useful in understanding CPUE estimates which are location-specific whilst providing additional biological information such as morphometric measurements and condition indices [20,36]. The CPUE for *C. pagurus* and *H. gammarus* was greater outside the Berwickshire Marine Reserve designation. This study did not focus on the potential "spill-over" effect associated with the Berwickshire Marine Reserve, but it may be suggestive based on the CPUE trend outside the designation. It can be assumed that the effort required to fish for *C. pagurus* and *H. gammarus* is less outside the marine reserve but much greater inside, suggesting overexploitation within the marine reserve. This is contrary to a similar study for the marine protected area (MPA) around the Isle of Arran, Scotland, in which the inverse was found [23]. Inside the reserve, the CPUE for *H. gammarus* was 123% higher within the designation in 2012; with *C. pagurus*, the CPUE was similar both inside and outside the designation [23]. However, in 2013 it was found that the CPUE of *C. pagurus* had declined within the reserve by 49%, equating to a 253% difference between the outside and inside areas of study [23]. Studies outside the U.K. also find increases in abundance inside the designated areas of protection which have also been related to a "spill-over" effect. Spiny lobsters (*Panulirus interruptus*) in California were 124% higher in abundance inside the reserve and 223% higher on the border of the reserve, suggesting a "spill-over" effect over a 10-year sampling period (2008–18) [37]. Furthermore, in Norway, a 245% increase in CPUE for *H. gammarus* was located inside MPAs based on pooled results from 2006 to 2010 [38]. It has been stated that warmer sea temperatures are likely to increase catch due to the metabolic demand of the *C. pagurus* species [20]. In this study, it was found that sea surface temperature correlated with the number of *C. pagurus* caught above the minimum landing size, which was also similar to that of [20]. This contrasts observations with *C. pagurus* fishing occurring typically between April and November, with the highest catches observed in October and November [20]. CPUE of *H. gammarus* also showed a negative correlation with sea surface temperature in this study, but [23] found that temperature had no significant effect on the number of individuals caught. In the Mediterranean Sea, warmer waters (May–August) were associated with higher CPUE for *H. gammarus* [39]. Soak time ranged from 1 to 7 days, with the optimal soak times differing depending on which target species was being caught. It has been suggested that soak time does not linearly affect the end catch [20], which is apparent in Figures 4–6. Other studies suggest that the number of *C. pagurus* landings positively correlates with soak time [36] but also declines with increasing soak time [39]. In the Northern Taiwan Strait, soak times of <48 h saw a greater catch of target crustacean species, whereas soak times of >48 h saw a greater haul of bycatch [40]. In this study, soak time had a significant effect on the CPUE for all target species as a smoothing factor in the GAM models, although the effects of soak time greatly differed depending on species (Figures 4–6). Depth also showed a significant effect on CPUE, except for *H. gammarus*, which has also been shown in [20] with the number of *C. pagurus* caught increasing with depth. However, it has been shown in [4] that the number of *C. pagurus* caught showed a nonlinear decrease with depth (23–84 m); however, this depth range is far deeper than that used in this study (5–37 m) [4]. Creel deployment is typically associated with rocky habitats compared to sand or muddy substrates [6]. Over a 9-year period, the fishing pressure in the rocky inshore area doubled in the Northumberland district, although the creel density was constant [6]. In areas of high fishing density, it has been shown that the number of *C. pagurus* and *H. gammarus* caught declined over a 3-year period by 19% and 35% respectively [41]. Future data surveys should integrate the habitat type on which the fishing equipment is deployed, based on the data provided in Figure 1, also known as fisheries habitat interactions [6].

Creels are not standardised fishing equipment; fishers can adapt and change them in response to fishing needs. Such adaptations can include frame material, net gap size, and number and size of entrances [20]. Predominantly, in this area, the creels are not affixed with escape gaps, which likely increases creel saturation. Such escape gaps can increase the efficiency of the creel by decreasing the undersized catch by 34% whilst increasing commercial catch by 125%, which is further exaggerated with the inclusion of two gaps [42]. Many studies record the efficiency of escape gaps and their ability to reduce the catch of undersized individuals whilst maintaining the commercial catch (those at MLS or above) [13,42–44]. The results in this study showed that CPUE and the number of target species caught greatly varied among the various creel types used. The CPUE of *N. puber* in 2019 was greater in the prawn parlour variations (Table 2), CPUE of *C. pagurus* was greater in parlour creels, and CPUE of *H. gammarus* was greater in hard- and soft-eye creels. From fisher observations, the hard-eye creel variations are dominated by *H. gammarus*, and the soft-eye creels are dominated by *C. pagurus*, although this cannot be confirmed by this study as the number of *H. gammarus* dominantly outnumbered *C. pagurus* in all creel types (Table 2). Other adaptations made to creels can include the entrances, with larger and smaller entrances potentially limiting sized individuals. The entrances for the creels used in this study were not measured, and assumptions on catch limitations by entrance size cannot be made and supported. Typically, the size of entrances on creels ranges from 3.5″ (89 mm) to 6″ (152 mm) in the Berwickshire area, although the industry standard is recorded as 9.8″ (250 mm) [29]. A study looking into the influence of entrance size on crayfish noted that the entrance size had no effect on the range of sizes caught [43]. The entrance sizes ranged from 45 to 85 mm in diameter [43], significantly smaller than those required to meet the landing sizes of both *C. pagurus* and *H. gammarus*. In contrast, it has been documented that the entrance size, if increased, is likely to increase the mean size of the animal caught, in this case, *H. gammarus* [13].

4.3. Bait

The dominant bait type used across all fishing vessels was mackerel (*Scomber scombrus*) (42.07%) and herring (*Clupea harengus*) (34.36%). Both *H. gammarus* and *C. pagurus* are chemosensory species using bimodal sensilla and olfactory systems to respond to chemical stimuli [9,45]. Both herring and mackerel are recognised as oily fish [46] and, it would be assumed, produce a strong chemical stimulus. The chemosensory stimulus is dependent on time, as the degradation and decomposition of the bait will influence the time period at which the fishing gear is most effective, with bait degradation ranging from 4 to 27 days based on values from [47,48]. It could be assumed that bait plumes surrounding the creel pots can vary regionally due to the regional variation in tides and currents [20]. The bait plume influence and the area with which the plume was associated could not be estimated in this study. Strong currents could lead to faster dilution of the bait plume, which could lead to smaller areas of influence that can only be estimated or assumed [9]. When presented with mackerel bait in a closed flume system, scampi (*Metanephrops challengeri*) showed no alterations in their behaviour but showed greater walking speeds in more turbulent water flow influenced by such bait [49]. Contrastingly, free-ranging *H. gammarus* exhibited a decrease in walking speed with decreasing distance to deployed creels; however, the behaviour could not be associated with bait influence specifically [9]. This chemosensory stimulus may influence a deterrence behaviour, as crushed crab added to baited creels led to a decrease in caught crabs by 54% [50].

4.4. Models

The best-fitting models for CPUE of *H. gammarus* and *C. pagurus* were those excluding the factors creel type and bait type. When additional explanatory factors were applied to each model, the deviance explained increased. However, more variables were significant, which seemed to exaggerate the interpretation of the models. The model used was based on that of [20], with the inclusion of bait type and creel type. Both smoothing parameters, soak time and depth, influenced the CPUE for all target species with varying influences. The CPUE of *C. pagurus* seemed to increase up to a period of 5 days before steadily decreasing, whereas that of *H. gammarus* decreased to 4 days before plateauing (Figures 5 and 6). Soak time varied depending on the fishing vessel used and the season. The soak time of static fishing gear can be influenced by the weather, with early termination of fishing periods being dependent on the adversity of the weather. If the creels are heavily saturated, this could likely lead to a decrease in available bait and space, further inhibiting catch. A higher density of *H. gammarus* in the creels will limit the catch of *C. pagurus* [2,51]; in contrast, a higher density of *C. pagurus* would not limit the available space but would reduce the available bait, limiting the attraction of other individuals [52]. The presence of one individual *H. gammarus* within a creel can decrease the CPUE of *C. pagurus* and *N. puber* by factors of 12 and 9, respectively [6]. Similar to the smoothing trend of soak time on the CPUE of *H. gammarus*, CPUE decreased with increasing depth until around 20 m before plateauing. The CPUE of *C. pagurus* increased with increasing depth, whilst that of *N. puber* increased with increasing depth until around 20 m before decreasing. The influence of depth on CPUE is likely linked to bottom sea temperature and sea current, which could be utilised to improve location-specific data [2,20]. *C. pagurus* are well known to migrate further offshore; in particular, females migrate further offshore during the gestation period to bury themselves in finer sediment such as sand for brooding [42,53]. It is documented that the depth range distribution of *C. pagurus* is around 6–40 m [54]. The depth range of *H. gammarus* ideally ranges from 35 to 40 m [40]. *H. gammarus* have expressed depth diel patterns from September to January as a form of inactivity coinciding with the decrease in water temperature [55,56]. The same study showed that all but one individual remained in depths shallower than 30 m for longer and shorter periods of activity [46]. The CPUE of *H. gammarus* was influenced by depths below 30 m (Figure 5) which would suggest a similar trend to [53], but also the limited activity in this species is likely to be associated with site fidelity which is common with this species [57], with activity commonly associated with feeding and territorial behaviour [58] within a few (~3.8 km) kilometres from their shelter [39]. Within a 20 m distance of a deployed creel, *H. gammarus* can remain in this vicinity for up to 17 h, and their presence can reduce the catch of both *C. pagurus* and *N. puber* [2,9].

5. Conclusions

Our study suggests that there is an exploitation of the commercially important species *C. pagurus* and *H. gammarus* inside the Berwickshire Marine Reserve due to the lower CPUE values recorded. A "spill-over" effect for both species may be a factor contributing to the varying CPUE values recorded in and outside the marine reserve, which would require further investigation. With no creel limitation in the region, it may be suggested that the decline inside the marine reserve may be accounted for by creel density saturating any viable fishing ground and subsequently contributing to regional fishing pressure. This would require further investigation through monitoring of vessel activity within the reserve designation gathered with autonomous tracking devices attached to the inshore fishing fleet. The depth and soak time of deployed creels were associated with the CPUE of the target species. This information could be used in relation to a creel limitation. By reducing the creel saturation on fishing grounds at certain depths and decreasing or increasing the length of species-dependent soak times, more targeted fishing activity can take place without affecting the overall catch for fishers. This is an ideal within a complex system, as other factors such as substrate type and bottom sea temperature would need to be considered, which would highlight fishery habitat interactions, improving the resolution of the findings in this study. It is believed that the information provided in this study will be used in future advice and consideration for changes in fisheries management for Berwickshire.

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Informed Consent Statement: Informed consent was obtained from all inshore fishers prior to the commencement of the study to allow the use of their vessel and the publication of the data collected.

Data Availability Statement: The data collected and analysed in this paper are not available publicly. Readers can inquire and request the raw data files from the first author (blair.easton@marinestation.co.uk).

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References

1. Pauly, D.; Christensen, V.; Guénette, S.; Pitcher, T.J.; Sumaila, U.R.; Walters, C.J.; Watson, R.A.; Zeller, D. Towards sustainability in world fisheries. *Nature* **2002**, *418*, 689–695. [CrossRef]
2. Skerritt, D.J.; Bannister, R.C.A.; Polunin, N.V.C.; Fitzsimmons, C. Inter- and intra-specific interactions affecting crustacean trap fisheries—Implications for management. *Fish. Manag. Ecol.* **2020**, *27*, 445–453. [CrossRef]
3. Mcveigh, K. 'Much of Scottish Crab and Lobster Is "Fish to Avoid", Says Sustainable Seafood Guide', The Guardian. Available online: https://www.theguardian.com/environment/2022/apr/05/much-of-scottish-crab-and-lobster-is-fish-to-avoid-says-sustainable-seafood-guide (accessed on 6 April 2022).
4. Mesquita, C.; Dobby, H.; Pierce, G.J.; Jones, C.S.; Fernandes, P.G. Abundance and spatial distribution of brown crab (*Cancer pagurus*) from fishery-independent dredge and trawl surveys in the North Sea. *ICES J. Mar. Sci.* **2020**, *78*, 597–610. [CrossRef]
5. Marine Scotland, 'Scottish Sea Fisheries Statistics 2019', The Scottish Government, 2020. Available online: https://www.gov.scot/publications/scottish-sea-fisheries-statistics-2019/ (accessed on 26 June 2023).
6. Stephenson, F.; Polunin, N.V.C.; Mill, A.C.; Scott, C.; Lightfoot, P.; Fitzsimmons, C. Spatial and temporal changes in pot-fishing effort and habitat use. *ICES J. Mar. Sci.* **2017**, *74*, 2201–2212. [CrossRef]
7. Breen, P.; Vanstaen, K.; Clark, R.W.E. Mapping inshore fishing activity using aerial, land, and vessel-based sighting information. *ICES J. Mar. Sci.* **2014**, *72*, 467–479. [CrossRef]
8. Kafas, A.; McLay, A.; Chimienti, M.; Scott, B.E.; Davies, I.; Gubbins, M. ScotMap: Participatory mapping of inshore fishing activity to inform marine spatial planning in Scotland. *Mar. Policy* **2017**, *79*, 8–18. [CrossRef]
9. Lees, K.J.; Mill, A.C.; Skerritt, D.J.; Robertson, P.A.; Fitzsimmons, C. Movement patterns of a commercially important, free-ranging marine invertebrate in the vicinity of a bait source. *Anim. Biotelemetry* **2018**, *6*, 8. [CrossRef]

10. Murray, L.; Seed, R. Determining whether catch per unit effort is a suitable proxy for relative crab abundance. *Mar. Ecol. Prog. Ser.* **2010**, *401*, 173–182. [CrossRef]
11. Stamatopoulos, C. 'Sample-Based Fishery Surveys: A Technical Handbook', FAO Fisheries Technical Paper, no. 425, p. 132. 2002. Available online: https://www.fao.org/3/Y2790E/Y2790E00.htm (accessed on 28 June 2023).
12. Harley, S.J.; Myers, R.A.; Dunn, A.; Thorson, J.T.; Fonner, R.; Haltuch, M.A.; Ono, K.; Winker, H.; Carruthers, T.R.; Walter, J.F.; et al. Is catch-per-unit-effort proportional to abundance? *Can. J. Fish. Aquat. Sci.* **2001**, *58*, 1760–1772. [CrossRef]
13. Addison, J.T.; Lovewell, S.R.J. Size composition and pot selectivity in the lobster (*Homarus gammarus* (L.)) and crab (*Cancer pagurus* L.) fisheries on the east coast of England. *ICES J. Mar. Sci.* **1991**, *48*, 79–90. [CrossRef]
14. Hart, P.J.B. Enlarging the shadow of the future: Avoiding conflict and conserving fish. In *Reinventing Fisheries Management*; Pitcher, T.J., Pauly, D., Hart, P.J.B., Eds.; Springer Netherlands: Dordrecht, The Netherlands, 1998; pp. 227–238. [CrossRef]
15. Woll, A.K.; van der Meeren, G.I.; Fossen, I. Spatial variation in abundance and catch composition of Cancer pagurus in Norwegian waters: Biological reasoning and implications for assessment. *ICES J. Mar. Sci.* **2006**, *63*, 421–433. [CrossRef]
16. Fogarty, M.J.; Addison, J.T. Modelling capture processes in individual traps: Entry, escapement and soak time. *ICES J. Mar. Sci.* **1997**, *54*, 193–205. [CrossRef]
17. Bennett, D.B. Factors in the life history of the edible crab (*Cancer pagurus* L.) that influence modelling and management. *ICES Mar. Sci. Symp.* **1995**, *199*, 89–98.
18. Tully, O.; Robinson, M.; O'Keefe, E.; Cosgrove, R.; Doyle, O.; Lehane, B. The Brown Crab (*Cancer pagurus* L.) Fishery: Analysis of the Resource in 2004–2005. *Fish. Resour.* **2006**, *4*, 48. Available online: https://bim.ie/wp-content/uploads/2021/02/bimNo,4,The,Brown,-,Crab,Cancer,pagurus,L,-,Fishery,Analysis,of,the,resource,in,2004-2005,.pdf (accessed on 17 July 2023).
19. Okamura, H.; Morita, S.H.; Funamoto, T.; Ichinokawa, M.; Eguchi, S. Target-based catch-per-unit-effort standardization in multispecies fisheries. *Can. J. Fish. Aquat. Sci.* **2018**, *75*, 452–463. [CrossRef]
20. Öndes, F.; Emmerson, J.A.; Kaiser, M.J.; Murray, L.G.; Kennington, K. The catch characteristics and population structure of the brown crab (*Cancer pagurus*) fishery in the Isle of Man, Irish Sea. *J. Mar. Biol. Assoc. United Kingd.* **2019**, *99*, 119–133. [CrossRef]
21. Pendleton, L.H.; Ahmadia, G.N.; Browman, H.I.; Thurstan, R.; Kaplan, D.M.; Bartolino, V. Debating the effectiveness of marine protected areas. *ICES J. Mar. Sci.* **2018**, *75*, 1156–1159. [CrossRef]
22. Sala, E.; Giakoumi, S. No-Take Marine Reserves Are theMost Effective Protected Areas in the Ocean. *ICES J. Ofmarine Sci.* **2017**, *75*, 1166–1168. [CrossRef]
23. Howarth, L.M.; Dubois, P.; Gratton, P.; Judge, M.; Christie, B.; Waggitt, J.J.; Hawkins, J.P.; Roberts, C.M.; Stewart, B.D. Trade-offs in marine protection: Multispecies interactions within a community-led temperate marine reserve. *ICES J. Mar. Sci.* **2017**, *74*, 263–276. [CrossRef]
24. Marine Scotland. *Consultation on Landing Controls for the Scottish Crab and Lobster Fisheries: Outcome Report*; The Scottish Government: Edinburgh, Scotland, 2017; ISBN 978-1-78652-769-1.
25. Bell, S.; Aird, C.; Blackadder, L.; James, M.; Mendo, T.; Colilles, A.M.; Woulters, J.; Strang, N. *Outer Hebrides Inshore Fisheries Pilot: Year One Report*; The Scottish Government: Edinburgh, Scotland, 2022; ISBN 978-1-80435-440-7.
26. The Scottish Government. Fisheries Management Strategy 2020 to 2030: Delivery Plan. Available online: https://www.gov.scot/publications/scotlands-fisheries-management-strategy-2020-2030-delivery-plan/documents/ (accessed on 26 June 2023).
27. RStudio Team. *RStudio: Integrated Development for R*; RStudio, PBC: Boston, MA, USA, 2020. Available online: http://www.rstudio.com/ (accessed on 13 May 2021).
28. Pennington, M.; Burmeister, L.-M.; Hjellvik, V. Assessing the Precision of Frequency Distributions Estimated from Trawl-Survey Samples. 2002. Available online: http://hdl.handle.net/1834/31044 (accessed on 28 June 2023).
29. Edwards, E. *The Edible Crab and Its Fishery in British Waters*; Fishing News Books Ltd.: Farnham, UK, 1979.
30. Woll, A.K.; Alesund, M. The Edible Crab: Biology Grading Hand-Ling Live Crabs. Handbook. 2006. Available online: File:///C:/Users/MDPI/Downloads/Handbook%20-%20Edible%20crab%202005%20-%20MF.pdf (accessed on 25 March 2022).
31. Hall, S.J.; Robertson, M.R.; Basford, D.J.; Fryer, R. Pit-Digging by the Crab Cancer pagurus: A Test for Long-Term, Large-Scale Effects on Infaunal Community Structure. *J. Anim. Ecol.* **1993**, *62*, 59–66. [CrossRef]
32. Aitken, A. Brown Crab and European Lobster Fisheries in the NWIFCA District the Use of Returns Data to Inform Management, 2018. Available online: https://www.nw-ifca.gov.uk/app/uploads/Agenda-Item-10-Annex-A-TSB-Annex-A-Crab-and-Lobster-Report-Use-of-Landings-Data-08-01-18.pdf (accessed on 27 June 2023).
33. Pawson, M.G. *Biogeographical Identification of English Channel Fish and Shellfish Stocks. Ministry of Agriculture, Fisheries and Food, Fisheries Research Technical Report*; MAFF Directorate of Fisheries Research: Lowestoft, UK, 1995; Volume 99, p. 72.
34. Kaiser, M.J. Directorate-General for Internal Policies Policy Department B: Structural and Cohesion Policies Fisheries the Conflict between Static Gear and Mobile Gear in Inshore Fisheries Study. 2014. Available online: https://www.europarl.europa.eu/RegData/etudes/STUD/2014/529070/IPOL_STU(2014)529070_EN.pdf (accessed on 27 June 2023).
35. The Scottish Government. Changes to Legislation: There Are Currently No Known Outstanding Effects for the Inshore Fishing (Scotland) Act 1984. 1984. Available online: https://www.legislation.gov.uk/ukpga/1984/26/contents (accessed on 28 June 2023).
36. Verdoit, M.; Pelletier, D.; Bellail, R. Are commercial logbook and scientific CPUE data useful for characterizing the spatial and seasonal distribution of exploited populations? The case of the Celtic Sea whiting. *Aquat. Living Resour.* **2003**, *16*, 467–485. [CrossRef]

37. Lenihan, H.S.; Fitzgerald, S.P.; Reed, D.C.; Hofmeister, J.K.K.; Stier, A.C. Increasing spillover enhances southern California spiny lobster catch along marine reserve borders. *Ecosphere* **2022**, *13*, e4110. [CrossRef]
38. Moland, E.; Olsen, E.M.; Knutsen, H.; Garrigou, P.; Espeland, S.H.; Kleiven, A.R.; Andre, C.; Knutsen, J.A. Lobsterand Cod Benefit from Small-Scale Northern Marine ProtectedAreas: Inference from an Empirical Before–After Control-Impact Study. *Proc. R. Soc. B Biol.* **2013**, *280*, 20122679. [CrossRef]
39. Pavičić, M.; Matić-Skoko, S.; Vrdoljak, D.; Vujević, A. Population Characteristics of the European Lobster, *Homarus gammarus* in the Adriatic Sea: Implications for Sustainable Fisheries Management. *Water* **2021**, *13*, 1072. [CrossRef]
40. Naimullah, M.; Lee, W.-Y.; Wu, Y.-L.; Chen, Y.-K.; Huang, Y.-C.; Liao, C.-H.; Lan, K.-W. Effect of soaking time on targets and bycatch species catch rates in fish and crab trap fishery in the southern East China Sea. *Fish. Res.* **2022**, *250*, 106258. [CrossRef]
41. Rees, A.; Sheehan, E.V.; Attrill, M.J. Optimal fishing effort benefits fisheries and conservation. *Sci. Rep.* **2021**, *11*, 3784. [CrossRef] [PubMed]
42. Brown, C.G. The effect of escape gaps on trap selectivity in the United Kingdom crab (*Cancer pagurus* L.) and lobster (*Homarus gammarus* (L.)) fisheries. *ICES J. Mar. Sci.* **1982**, *40*, 127–134. [CrossRef]
43. Brown, P.; Hunt, T.L.; Giri, K. Effects of gear type, entrance size and soak time on trap efficiency for freshwater crayfish Cherax destructor and C. albidus. *Mar. Freshw. Res.* **2015**, *66*, 989–998. [CrossRef]
44. Winger, P.D.; Walsh, P.J. Selectivity, efficiency, and underwater observations of modified trap designs for the snow crab (*Chionoecetes opilio*) fishery in Newfoundland and Labrador. *Fish. Res.* **2011**, *109*, 107–113. [CrossRef]
45. Hallberg, E.; Skog, M. *Chemosensory Sensilla in Crustaceans*; Springer: Berlin/Heidelberg, Germany, 2011; pp. 103–121. [CrossRef]
46. Food Standards Agency. What's an Oily Fish? The National Archives, 24 June 2004. Available online: https://www.food.gov.uk/business-guidance/hsc-nutritional-standards-proteins (accessed on 28 June 2023).
47. Bullimore, B.A.; Newman, P.B.; Kaiser, M.J.; Gilbert, S.E.; Lock, K.M. A Study of Catches in a Fleet of "Ghost-Fishing" Pots. *Fish. Bull.* **2001**, *99*, 247.
48. Putsa, S.; Boutson, A.; Tunkijjanukij, S. Comparison of ghost fishing impacts on collapsible crab trap between conventional and escape vents trap in Si Racha Bay, Chon Buri province. *Agric. Nat. Resour.* **2016**, *50*, 125–132. [CrossRef]
49. Major, R.; Jeffs, A. Orientation and food search behaviour of a deep sea lobster in turbulent versus laminar odour plumes. *Helgol. Mar. Res.* **2017**, *71*, 9. [CrossRef]
50. Chapman, C.J.; Smith, G.L. Creel catches of crab, *Cancer pagurus* L. using different baits. *ICES J. Mar. Sci.* **1978**, *38*, 226–229. [CrossRef]
51. Watson, W.; Jury, S.H. The relationship between American lobster catch, entry rate into traps and density. *Mar. Biol. Res.* **2013**, *9*, 59–68. [CrossRef]
52. Rayner, G.; McGaw, I.J. Effects of the invasive green crab (*Carcinus maenas*) on American lobster (*Homarus americanus*): Food acquisition and trapping behaviour. *J. Sea Res.* **2019**, *144*, 95–104. [CrossRef]
53. Howard, A.E. The Distribution and Behaviour of Ovigerous Edible Crabs (*Cancer pagurus*), and Consequent Sampling Bias. 1982. Available online: http://icesjms.oxfordjournals.org/ (accessed on 28 June 2023).
54. Tonk, L.; Rozemeijer, M. Ecology of the Brown Crab (*Cancer pagurus*) and Production Potential for Passive Fisheries in Dutch Offshore Wind Farms, p. 49. 2019. Available online: http://library.wur.nl/WebQuery/wurpubs/553352 (accessed on 28 June 2023).
55. Moland, E.; Olsen, E.M.; Andvord, K.; Knutsen, J.A.; Stenseth, N.C. Home range of European lobster (*Homarus gammarus*) in a marine reserve: Implications for future reserve design. *Can. J. Fish. Aquat. Sci.* **2011**, *68*, 1197–1210. [CrossRef]
56. Smith, I.; Collins, K.; Jensen, A. Seasonal changes in the level and diel pattern of activity in the European lobster *Homarus gammarus*. *Mar. Ecol. Prog. Ser.* **1999**, *186*, 255–264. [CrossRef]
57. Bannister, R.C.A.; Addison, J.T.; Lovewell, S.R.J. Growth, Movement, Recapture Rate and Survival of Hatchery Reared Lobsters (*Homarus gammarus* (Linnaeus, 1758)) Released into the Wild on the English East Coast. *Crustaceana* **1994**, *70*, 156–172. [CrossRef]
58. Karavanich, C.; Atema, J. Individual recognition and memory in lobster dominance. *Anim. Behav.* **1998**, *56*, 1553–1560. [CrossRef]

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Review

Marine Endangered and Threatened Species in Russia: A Review of Current Conservation Strategies and Management Legislative Tools

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Abstract: Despite the global decline in marine species biodiversity, relatively few countries have enacted national endangered and threatened species legislation. Tailoring an adequate legislative framework with clear objectives and regulations consistent with the available scientific evidence is fundamental for the effective conservation of marine endangered and threatened species. This paper analyzes the legal framework and current institutional tools for the conservation of marine endangered and threatened species in the Russian Federation. In this regard, important legislative tools include federal laws, as well as internationally binding signed agreements, among others, such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Convention on Wetlands of International Importance Especially as Waterfowl Habitat (Ramsar). A strategy and an action plan for the conservation of biological diversity were also developed. Besides, the most important tool for the conservation and protection of marine endangered and threatened species is the Red Book of the Russian Federation (RBRF) and other regional Red Books. Responsibility for causing harm to the species listed in the RBRF and their habitat is specified in the code of administrative offenses and the criminal code of the Russian Federation. Finally, conclusions and identified gaps were highlighted in the last section, including, among other things, that legislation is still limited in how it takes the impacts of climate change into account. Such type of study is highly recommended, considering the relatively few number of papers dedicated to the study of the impact and/or implications of the conservation tools and strategies mentioned in this paper on the status of the marine endangered and threatened species.

Keywords: endangered species; conservation; legislation; policy; Russia; threatened species

Key Contribution: Important conservation tools are being used in the Russian Federation for the protection of endangered and threatened species; although some gaps were identified.

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1. Introduction

Over the last decades, marine biodiversity has declined considerably as the number of marine endangered and threatened species is now higher (more species are threatened) than at any other time in recent history [1–4]. The degradation of the global marine ecosystems due to anthropogenic impacts such as pollution [5], habitat loss and degradation [6], overfishing [7], and climate change [8,9] increased the risk of extinction of many marine species. Some evaluations highlighted the high level of extinction risk in certain marine taxonomic groups, including 33% of reef-building corals [10], at least 25% of sharks and rays [11], 16% of mangroves [12], 14% of seagrasses [13], and 11% of billfish and scombrids (e.g., tunas, bonitos, mackerels) [14]. For these reasons, great attention has been paid to the conservation and management of marine species to halt the recent decline in their abundance and diversity and to preserve the ecosystem. To reduce biodiversity loss, many attempts and conservation actions were made by conservation biologists such as species and habitat protection, ex-situ programs, removing invasive species, education and awareness campaigns, and designing adequate measures for the conservation of these species [15]. Over the last five decades, the Red List of Threatened Species of the International Union for Conservation of Nature (IUCN) has been guiding conservation endeavors as a valuable species extinction risk assessment tool. It is widely assumed that such conservation endeavors and tools have been useful in sustaining species from moving closer to extinction and driving recoveries with numerous successful cases in which species have been brought back from the brink of extinction. Lotze et al. [16] provided some examples of population and ecosystem successful cases of recoveries. Yet some studies still define a new focus of recovery and seek to present frameworks for quantifying measures of species recovery and conservation success (e.g., [17]).

Even though the conservation of marine species and ecosystems is still often portrayed as a scientific duty it is important to acknowledge that science is fundamental but, on its own, is not the only component of the decision-making process. In the end, the decision 'to conserve' and/or 'to establish recovery plans' does not depend only on science, but rather, among other things, on tailoring an adequate legislative framework with clear objectives, regulations and monitoring plans that may or may not be consistent with the available scientific evidence. Therefore, the effective conservation of marine endangered and threatened species, as a first step, depends on legislation properly designed to protect them. Similarly, achieving the objectives of these legislative tools depends on the political and social factors that affect its implementation (e.g., [18]).

Despite the global decline in marine species biodiversity, relatively few countries have enacted national endangered and threatened species legislation. The first legislative act was established in the United States, with the Endangered Species Preservation Act in 1966, the forerunner of the Endangered Species Act (ESA; passed in 1973). This was been followed by other legislative tools in some countries such as; the Biodiversity Law of Costa Rica (passed in 1992), Australia's Endangered Species Protection Act (passed in 2002), Canada's Species at Risk Act (SARA; also passed in 2002), also in Canada: Nova Scotia Endangered Species Act (NSESA; passed in 1998), and South Africa's National Environmental Management Biodiversity Act (passed in 2004). Several studies have evaluated the effectiveness of some of these acts for preventing the extinction of listed species, with the ESA and SARA the most evaluated acts. For example, some studies evaluated the listing, designation of critical habitat and development of recovery plans, and improvement of species' status [19–24]. Other works have highlighted, for example, the temporal and policy-driven differences in the listing rate [25], biases in the listing of species towards particular taxonomic groups [26,27], and fluctuations in the funding allocated to ESA implementation [19]. However, recent comprehensive analysis is absent in many countries. To date, there have been limited efforts to evaluate the efficacy of similar legal frameworks in many countries, partially because these legislations often lack objective, quantifiable criteria to use in such assessments.

In Russia, particular attention has been paid to the conservation of marine endangered and threatened species. Recently in 2014, the government of the Russian Federation approved the strategy for the conservation of endangered and threatened species for the period until 2030. Despite the great importance of marine biodiversity in Russia, there are relatively few papers in this regard mostly in the Russian language [28–33]. Taking into account that little is known outside the country, this paper elucidates and analyzes the legal framework, current institutional instruments and strategies for the conservation of marine endangered and threatened species and reveals gaps in the Federal legal frameworks in the Russian Federation. We reviewed available legislation and laws to assess the current regulation practices in the country's conservation of marine endangered and threatened species. This would strengthen the knowledge about Russia's policy in the conservation of marine endangered and threatened species, and improve the legislation currently used as a conservation tool. In addition, the information gathered in this paper would be very important at the global level for the scientific community outside of the country given Russia's massive borders and presence on multiple shared seas (e.g., Barents Sea shared with Norway) and oceans.

2. General Methodology

This paper draws on findings from a literature review of official reports, government documents, legal reports, official governmental websites and unpublished sources relating to endangered and threatened species. The review provides the basis for a description of the key organizations, laws, decrees, objectives, and strategies that guide the conservation of endangered and threatened species. To obtain relevant data on Russian legislation, documents issued by the central Federal government, relevant ministries (e.g., Ministry of Natural Resources of Russia) and regional governments (e.g., Ministries of Natural Resources and Environment of the constituent entities of the Russian Federation) websites were reviewed. Most of the legislative documents issued in the Russian Federation are now available online on the websites of the management authorities (e.g., ministries), although mostly in Russian language. Besides, online platforms such as https://docs.cntd.ru and http://www.consultant.ru (accessed in 19 September 2022) (online platforms to disseminate many of the laws and regulations managed by the Russian government) were used. Terms in the Russian language such as 'strategy', 'law', 'regulation', 'fisheries', 'species', 'protected', 'endangered', and 'threatened' were combined and used to find and capture the main message of the relevant law articles and clauses within the collected policy and legislative documents. The detailed data analysis methodology of this study is presented in Figure 1. To answer the research questions, the review summarized the core aspects (content analysis) of each legal document and analyzed whether it provides effective protection to marine endangered and threatened species. The findings were compiled and presented in the following sections for comprehensive understanding and interpretation. Finally, conclusions and gaps identified from the review section were highlighted in the last section.

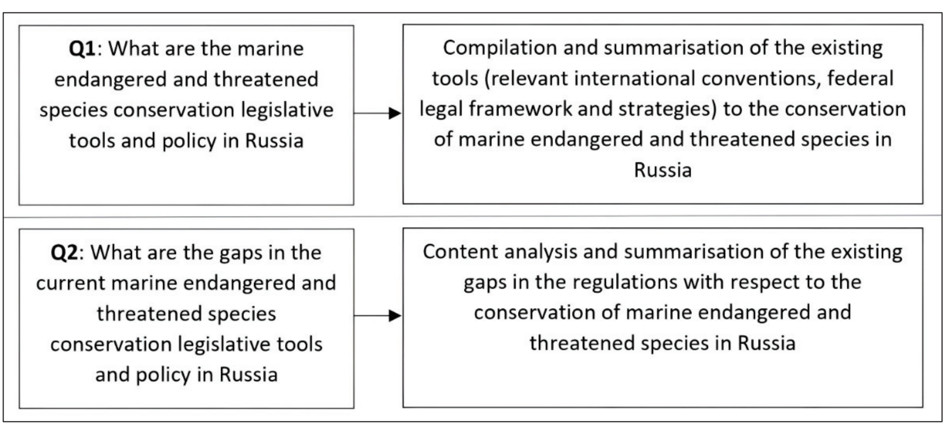

Figure 1. Schematic overview of the analysis made in this study.

3. Legal Framework for the Conservation of Endangered and Threatened Species

In the Russian Federation, the protection of endangered and threatened species (in Russian documents referred to as Rare and Endangered Species–herein after RES) is carried out within the framework of general legislation regulating the conservation of these species. The definition of "marine" or "aquatic" animals is given by the Federal Law "On fisheries and the conservation of aquatic biological resources". The main provisions for the protection of rare species are contained in the Federal Laws "On environmental protection" and "On the animal world". The legal act established for the purpose of protecting and listing RES is the Red Book of the Russian Federation. The regional Red Books are also adopted by each region of the Russian Federation separately. Further, the general and special regulatory legal acts of the Russian Federation, in force at the time of writing this paper, concerning the protection of RES in general, and which are also applicable to the protection of rare and endangered marine species and birds, are summarized in the following Table 1.

Table 1. Summary of the main Federal laws (listed by the issuing year) and related articles that contain the main provisions for the protection of rare species.

Legal Framework	Year	Related Articles	Core Objective
Federal law "On the animal world" [34]	1995	Article 1	Defines the wildlife in the exclusive economic zone (EEZ) of the Russian Federation.
		Article 16	Provides the federal executive bodies the authority of supervision in the field of protection and use of wildlife and their habitats aiming at preventing, detecting, and suppressing violations in use of wildlife and their habitats.
		Article 22	Indicates that any economic activity must take measures to preserve the habitat of wildlife and the conditions for their reproduction, feeding, rest and migration routes, and any change of these must be carried out in compliance with the requirements that ensure the protection of the animal world.
		Article 24	Regulates the procedure for including RES in the Red Book of the Russian Federation [35] and/or the regional Red Books.
Federal law "On specially protected natural areas" [36]	1995	Article 1	Defines Specially Protected Natural Areas (SPNA).
		In general	This law determines the status and categories of SPNA, the procedure for their creation and the regime of special protection.
Federal law No. 7-FZ "On environmental protection" [37]	2002	Article 3	Lists the conservation of biological diversity among the basic principles of environmental conservation.
		Article 60	Sets the Red Book of the Russian Federation [35] and the regional Red Books as the main tools for the protection and record of RES and prohibits any activity leads to its withdrawal or reduction.
Federal law No. 166-FZ "On fishing and conservation of aquatic biological resources" [38]	2004	Article 27	Prohibits the extraction of RES of aquatic biological resources (ABR).
		Article 50.1	States that the protection of RES of ABR is carried out in accordance with the Federal Law of January 10, 2002 No. 7-FZ "On environmental protection" and this Federal Law.

4. National Strategies for Biodiversity and RES Conservation

In accordance with the Decree of the President of the Russian Federation of April 19, 2017 N 176 "On the Strategy for the Environmental Security of the Russian Federation for the period up to 2025", the conservation of biological diversity is the main activity of Russia in the development of international cooperation to preserve, protect and restore the Earth's ecosystems [39].

The first national strategy for the conservation of biological diversity in Russia was adopted in 2001, and its main goal was formulated as follows: "Conservation of the biodiversity of natural biosystems at a level that ensures their sustainable existence and sustainable use". The provisions of the Strategy corresponded to the main provisions of the international agreement—the Convention on Biological Diversity. In 2010, the 10th Conference of the Parties to the Convention on Biological Diversity approved the Strategic Plan for the Conservation and Sustainable Use of Biodiversity for 2011–2020 and the Aichi Biodiversity Targets (Decision X/2) [40]. Based on the structure of this Strategic Plan and in accordance with the Aichi Targets, national targets were justified and formulated and the Action Plan for the conservation of biological diversity of the Russian Federation (2014) was developed [41]. This strategy and action plan also includes some national targets related to marine RES such as The Global Target 6 (sustainable harvesting of aquatic biological resources, reducing adverse impacts on threatened species and vulnerable ecosystems), The Global Target 10 (combating threats to the biological diversity of the seas) and The Global Target 12 (protection of endangered species). A set of indicators related to the number of different categories of protected species was provided to assess the achievement of the last objective.

In addition to the strategy for the conservation of biological diversity in Russia, a strategy for the conservation of RES of animals, plants and fungi in the Russian Federation for the period up to 2030 was developed (Decree of the Government of the Russian Federation of February 17, 2014 N 212-r "On Approval of the Conservation Strategy rare and endangered species of animals, plants and fungi in the Russian Federation until 2030") [42]. The provisions of this strategy define the objectives and main directions of state policy and activities in the field of conservation of RES necessary to improve the efficiency of public administration in this field. The purpose of this strategy is to ensure, on a long-term basis, the conservation and restoration of RES in the interests of the sustainable development of the Russian Federation. The main measures for the conservation of RES, provided for by this Strategy, include: (1) improvement of the system of state management and supervision in the field of protection and use of all species and their habitat; (2) improvement of the regulatory legal framework; (3) ensuring the continuity and systematic maintenance, regular updating of the Red Book of the Russian Federation and the Red Books of the constituent entities of the Russian Federation; (4) ensuring the functioning of an effective system of SPNA; (5) creation of a single federal centre that monitors, maintains a cadastre of all animal species in the format of a multi-level information system that provides for the rapid collection and analysis of incoming information from all over the Russian Federation and the subsequent provision of this data to interested parties; (6) ensuring the fulfilment of the obligations of the Russian Federation arising from international conventions and agreements, as well as Russia's membership in international organizations; (7) ensuring openness of information about the state of RES of animals, plants and fungi and their habitats, as well as about the measures taken for their protection and reproduction.

5. Red Book of the Russian Federation

Besides the previously mentioned laws and strategies, the most important tool for the conservation and protection of RES is the Red Book of the Russian Federation (hereinafter referred to as RBRF). It was published for the first time in August 1978, its release was timed to coincide with the opening of the XIV General Assembly of the International Union for Conservation of Nature (IUCN), held in the USSR. The latest and current edition of the RBRF was issued in 2020 and is published at least once every 10 years. The legal framework is the order of the Ministry of Natural Resources of Russia dated May 23, 2016 N 306 (as amended on 5 July 2021) "On Approval of the Procedure for Maintaining the Red Book of the Russian Federation" [43]. The RBRF is an official document containing a set of information on the state, distribution, categories of rarity status and endangered status as well as the protection measures required to ensure the conservation and recovery of RES (subspecies, populations) of wild animals, wild plants and fungi living on the territory (including aquatic area), the continental shelf and in the exclusive economic zone of the Russian Federation [35].

Currently, the RBRF is maintained by the Ministry of Natural Resources of Russia through a specially created commission on rare and endangered animals, plants and fungi. The commission interacts with scientific organizations, including the Russian Academy of Sciences (RAS), as well as with federal executive authorities and executive authorities of the constituent entities of the Russian Federation exercising powers in the field of protection and use of wildlife, including ABR. The Ministry of Natural Resources of Russia considers proposals for inclusion into or exclusion from the RBRF or for changing the categories of the status of species received from state authorities, organizations, and citizens, and sends these proposals to the RAS and, if necessary, to other scientific organizations to obtain their opinions. After the submission of expert opinions from the RAS and/or other scientific organizations, the Ministry of Natural Resources of Russia sends them, together with the indicated proposals, to the Commission for consideration. The inclusion into or exclusion from the RBRF or for changing the categories of RES is based on the published scientific data and monitoring of these species (scientific estimates of the number of species), as well as on the degree of vulnerability and threat of reduction in its number and/or range, an increase in the fragmentation of the range, on adverse changes in the conditions for the existence of this species or other data indicating the need to adopt special measures for its conservation and restoration. After the review of the commission, the Ministry of Natural Resources of Russia places all proposals as well as supporting scientific data on its official website at least 180 calendar days before making an appropriate decision.

Species of flora and fauna listed in the RBRF include one of the rarity status categories: 0—Probably extinct, 1—Endangered, 2—Decreasing in number and/or distribution, 3—Rare, 4—Uncertain in status, 5—Recoverable and recovering. Also, one of the threat of extinction status categories characterizing their state in their natural habitat: Disappeared in the wild (EW—Extinct in the Wild), Disappeared in the Russian Federation (RE—Regionally Extinct), Critically Endangered (CR—Critically Endangered), Endangered (EN—Endangered), Vulnerable (VU—Vulnerable), Near Threatened (NT—Near Threatened), Causing the least concern (LC—Least Concern), Insufficient data (DD—Data Deficient). Finally, to one of the categories of the degree and priority of environmental measures taken and planned for adoption (nature conservation status): I priority—immediate adoption of comprehensive measures is required, including the development and implementation of a conservation strategy and/or a program for the restoration (reintroduction) of species, II priority—it is necessary to implement one or more special measures for the conservation of species, III priority—enough general measures provided for by the regulatory legal acts of the Russian Federation in the field of environmental protection, organization, protection and use of SPNA and protection and use of the animal world and its habitat, for the conservation of species listed in the RBRF.

The RBRF consists of two volumes, the first volume is devoted to animals, and the second to plants. The volume dedicated to animals consists of two parts: invertebrates and vertebrates. In turn, vertebrates are considered in six sections: lampreys, ray-finned fish, amphibians, reptiles, birds and mammals.

When it comes to fish species, in the latest edition of the Red Data Book (2021), they are represented by seven orders: sturgeons (Acipenseriformes), scads (Clupeformes), salmons (Salmoniformes), cyprinids (Cypriniformes), cods (Gadiformes), eels (Anguilliformes) and sticklebacks (Gasterosteiformes). The most numerous protected fish species are sturgeons (nine species), salmonids (17 species, 6 subspecies, 4 populations) and cyprinids (seven species, one subspecies, two populations). Herring-like species are represented by two species, cod-like, eel-like and stickleback-like contain one species each.

Most sturgeon species have one category of rarity—a species that is on the verge of extinction and has the highest priority of conservation measures. Two species—the European (Atlantic) sturgeon (*Acipenser sturio*) and the Baltic sturgeon (*Acipenser oxyrinchus*) have a rarity category 0, that is, they belong to populations that have probably disappeared in Russia. The rest have a second category of rarity—a species that is declining in number or distribution.

Salmonids are represented by fish of 1, 2 and 3 categories of rarity in similar proportions, which reflects their status according to the IUCN. Herrings have 2 and 3 (rare species) categories of rarity. Most cyprinids are represented by fish with the second category of rarity, that is, they are declining in number or distribution. Cod-like and eel-like are the first category of rarity, and stickle-like—the second.

Thus, fish listed in the RBRF are represented by species with a variety of types of aquatic habitats: from rivers and lakes to marine areas. On the other hand, Regional Red Data Books have a greater number of fish species, depending on the status of their populations in the territories of a particular subject of the Russian Federation.

Species listed in the RBRF are completely withdrawn from economic use. Removal from nature is allowed only with the permission of specially authorized state bodies and only in special cases (see Section 9: Exceptions to restriction on harvesting RES). Fees and methods of calculation for accidental or intentional damage caused to them and their habitats have been approved for almost all RBRF species.

6. International Agreements for the Protection of Aquatic RESs and Birds

The Russian Federation signed a number of international conventions and framework agreements for the protection of RES of ABR and birds (summarized later in Table 3).

7. Federal State Supervision in the Field of Protection of RES

The main federal government agencies for the protection, control, and regulation of the use of wildlife and their habitats in the Russian Federation are the Federal Service for Supervision of Natural Resources (Rosprirodnadzor) and the border agencies of the Federal Security Service of the Russian Federation.

The powers to protect RES are vested in the Federal Service for Supervision of Natural Resources (Rosprirodnadzor) (in relation to the RBRF) and the subjects of the Russian Federation (in relation to regional Red Books). For example, the Far Eastern Interregional Directorate of Rosprirodnadzor exercises the following powers: (1) federal state control (supervision) in the field of protection, reproduction and use of wildlife and their habitats in SPNA of federal significance that are not under the control of federal state budgetary institutions; (2) federal state control (supervision) in the field of protection and use of SPNA in areas of federal significance and within the boundaries of their protected areas that are not under the control of federal state budgetary institutions; (3) federal state control (supervision) in the field of handling animals, with the exception of handling service animals, in terms of compliance with the requirements for the maintenance and use of wild animals kept or used in captivity, including those belonging to species listed in the RBRF and/or protected by international treaties of the Russian Federation (with the exception of compliance with the requirements for the maintenance and use of such animals for cultural and entertainment purposes); (4) protection of ABR listed in the RBRF, with the exception of ABR located in SPNA of federal significance.

In addition, Federal state control (supervision) in the field of fisheries and the conservation of marine ABR is carried out by the border agencies of the Federal Security Service of the Russian Federation. General supervision over the implementation of legislation is carried out by the prosecutor's office.

8. Liability for the Damage Caused to RES

Responsibility for causing harm to species listed in the RBRF and their habitats is described in Article 8.35 of the code of administrative offenses of the Russian Federation: "Destruction of rare and endangered species of animals or plants" [44] (Table 2). It is also specified in Article 259 of the criminal code of the Russian Federation: "Destruction of critical habitats for organisms listed in the Red Book of the Russian Federation" [45] (Table 2). In addition, Article 258.1 of the criminal code of the Russian Federation provides for separate liability for the illegal extraction and trafficking of especially valuable wild animals and ABR belonging to species listed in the RBRF and/or protected by international treaties of the Russian Federation. The list of such species is established by a Decree of the Government of the Russian Federation of October 31, 2013 N 978 which also includes anadromous fish species, including Atlantic sturgeon (*Acipenser sturio*), kaluga (*Huso dauricus*), Sakhalin sturgeon (*Acipenser mikadoi*), and Sakhalin taimen (*Parahucho perryi*).

Table 2. Summary of the main legal tools related to the responsibility for causing harm to species listed in the RBRF and their habitats and the corresponding sanctions.

Legal Framework	Article	Actions	Author	Sanction/Fine
Code of administrative offenses of the Russian Federation	Article 8.35	Actions (inaction) that can lead to death, reduction in the number of RES listed in the RBRF or violation of their habitats. Also, extraction, storage, transportation, collection, maintenance, sale or transfer of RES, their products, parts, or derivatives without a proper permit.	Citizens	Administrative fine: 2500 to 5000 rubles with or without confiscation of tools.
			Officials	Administrative fine: 15,000 to 20,000 rubles with or without confiscation of tools
			Legal entities	Administrative fine: 500,000 to 1 million rubles with or without confiscation of tools
Criminal code of the Russian Federation	Article 258.1	Illegal extraction and trafficking of RES listed in the RBRF and/or protected by international treaties of the Russian Federation		Fine: 300,000 to 500,000 rubles or in the amount of the wage or other income of the convicted person for a period of two to three years. Compulsory labor: from four hundred and eighty hours to three years. Prison: up to three years.
	Article 259	Destruction of critical habitats for RES listed in the RBRF, which cause the extinction of their populations.		

Illegal production, maintenance, acquisition, storage, transportation, shipment and sale of especially valuable wild animals and ABR belonging to species listed in the RBRF and/or protected by international treaties of the Russian Federation, their parts and derivatives, are punished by compulsory, corrective or forced labor with a fine up to imprisonment. Similar actions committed using the mass media or the internet, or by a person using his official position; either with a public demonstration, or by a group of persons by prior agreement or by an organized group, lead to stricter criminal liability.

To sum up all the previously mentioned tools, apart from the Federal laws (mentioned in Table 1), Table 3 summarizes the main Federal conservation tools and international agreements and conventions for the protection of rare species.

Table 3. Summary of the main Federal conservation tools and international agreements and conventions for the protection of rare species.

Scope	Category	Legal Framework	Year	Core Objective
National Conservation tools	Strategy for the conservation of RES for 2030	Decree of February 17, 2014 N 212-r	2014	To ensure, on a long-term basis, the conservation and restoration of RES of animals, plants and fungi in the interests of the sustainable development of the Russian Federation.
	Strategy for the Environmental Security of the Russian Federation for the period up to 2025	Decree of April 19, 2017 N 176	2017	Includes some national targets related to marine RES (e.g., the Global Targets 6, 10 and 12).
	Red Book of the Russian Federation	Order of the Ministry of Natural Resources, May 23, 2016 N 306 (as amended on 5 July 2021)	2021	The official document containing a set of information on the state, distribution, categories of rarity status and endangered status as well as the protection measures required to ensure the conservation and recovery of RES in the EEZ of the Russian Federation.
	Liability for the damage caused to RES	Article 8.35 of the code of administrative offenses		Sanctions for causing harm to species listed in the RBRF and their habitats.
		Article 259 and Article 258.1 of the criminal code of the Russian Federation		Sanctions and penalties for causing harm to species listed in the RBRF and their habitats.
International agreements and conventions for the protection of aquatic RESs and birds	Convention on Wetlands of International Importance, Mainly as a Habitat for Waterfowl (Ramsar)		1971	Conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world
	Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)		1973	To ensure that international trade in specimens of wild animals and plants does not threaten the survival of the species
	Convention on the Conservation of Migratory Species of Wild Animals (CMS)		1979	Provides a global platform for the conservation and sustainable use of migratory animals and their habitats
	Framework Convention for the Protection of the Marine Environment of the Caspian Sea (Tehran convention)		2003	Lays down the general requirements and the institutional mechanism for environmental protection in the Caspian region.

9. Exceptions to Restriction on Harvesting RES

In exceptional cases, the legislation of the Russian Federation allows the extraction of rare species of ABR. The resolution of the Decree of the Russian government of December 24, 2008 N 1017 “On the extraction (catch) of rare and endangered species of aquatic biological resources" together with the "Rules for the extraction (catch) rare and endangered species of aquatic biological resources", among other things, concerns species listed in the RBRF [46]. According to this resolution, the extraction of rare species of ABR is allowed for the following purposes: (1) conservation of ABR; (2) monitoring the state of their populations; (3) implementation of their artificial reproduction or acclimatization; (4) ensuring the maintenance of a traditional way of life and the implementation of traditional economic activities of the indigenous peoples of the North, Siberia and the Far East of the Russian Federation.

It should be noted that the extraction of the RES of ABR listed in the RBRF is carried out on the basis of permits issued by the Federal Service for Supervision of Natural Resources in the manner determined by the Ministry of Natural Resources and Ecology of the Russian Federation. Meanwhile, the other RES of ABR (i.e., not listed in the RBRF) such extraction is carried out on the basis of permits issued by the Federal Fisheries Agency. The rules establish that the tools, gears and methods used for the extraction of rare ABR should not cause damage to the natural populations of these ABR and their habitat. Also, the tools, gears and methods of extraction used must be selective and in the case of harvesting marine mammals in a living form, to reduce the possibility of causing them any physical and mental trauma. The control over compliance with these rules is carried out by both the Federal Service for Supervision of Natural Resources and the Federal Agency for Fisheries.

10. Conclusions and Identified Gaps

This is the first attempt to summarize the main legislative tools and policies related to endangered and threatened species in the Russian Federation. Marine legislation has been developing for many decades and is the key means by which the conservation and protection of marine biodiversity including endangered and threatened species is achieved. In addition, recently an increasing focus on 'holistic' policy development is evident (e.g., strategy and action plan for the conservation of biological diversity of the Russian Federation) compared with earlier 'piecemeal' sectoral approaches. Important marine legislative tools being used in the Russian Federation for the protection and conservation of endangered and threatened species, including Federal laws "On fishing and conservation of aquatic biological resources", "On environmental protection" and "On the animal world", as well as internationally binding signed agreements, among others, such as CITES and Ramsar. Besides the previously mentioned laws and strategies, the most important tool for the conservation and protection of marine endangered and threatened species is the RBRF and other regional Red Books. Responsibility for causing harm to the species listed in the RBRF and their habitat is specified in the code of administrative offenses and also the criminal code of the Russian Federation.

Some studies are devoted to comparing the effectiveness of the existing regional, national and international Red Books and red lists on the example of certain regions of the Russian Federation (e.g., [47,48]). For example, a comparative analysis of the Red Books and red lists applicable in St. Petersburg and its district showed that currently, these official documents are so different from each other that it can be difficult to combine them into one database. It was shown that in some cases the conservation status turns out to be purely formal, while in reality red-listed species are not always properly protected from the negative impacts of human activities. It is necessary to bring into a single system the conceptual gradation of categories of rarity vulnerability of the protected species and to define clear rules for compiling their lists. The general unification of red lists could significantly increase the effectiveness of the decision-making process in the field of the protection of rare and endangered species [48].

Most of the relevant Russian scientific literature in the last 15 years, is focused on the review of the distribution and dynamic of RES in Russian waters (e.g., at the regional level [49–52]) and/or proposals for the inclusion of some species in the upcoming edition of the RBRF (e.g., [53,54]). However, there are very limited studies that analyzed the impact or implications of the conservation tools and strategies (e.g., SNPA or RBRF) mentioned in this paper on the status of the RES. For example, Sereda (2016) presented how the Taganay

National Park contributes to the study of RES included in the Red Book of the Chelyabinsk Region, providing some examples of fish species, including grayling *Thymallus thymallus*, brown trout *Salmo trutta*, Siberian taimen *Hucho taimen*, European bullhead *Cottus gobio* [55]. Similarly, Dolganov and Tyurin (2014) provided information on the Far Eastern Marine Biosphere Reserve in Peter the Great Bay, Sea of Japan, indicating that about 500 vertebrates, among them 184 fish species were recorded in the reserve and highlighting that, over the last 20 years, the number of fish species was increased about 14 species [56]. Another example of analyzing some of the mentioned conservation tools is Zhevlakov's (2014) article which analyzed the problems of the legislative structure and application of Art. 258.1 of the Criminal Code of the Russian Federation about the illegal harvesting and trafficking of RES listed in the RBRF [57]. The author draws his conclusions and makes suggestions to improve this Article and its practical application, differentiating between similar administrative offenses. Such a type of study is highly recommended, considering the relatively few number of papers dedicated to the study of the impact or implications of the conservation tools and strategies mentioned in this paper on the status of the RES.

This review also showed a number of gaps and issues in the legislation related to the protection of RES in Russia. One of the main identified gaps is that the legal status of protection of species listed in the RBRF does not depend on the categories of rarity established for them in the RBRF itself. The national legislation does not provide for the differentiation of the level of protection of species and their habitats listed in the RBRF on the grounds of their rarity and specific threats of extinction: whether the species is endangered or relatively safe and recovering, their degree of protection is the same, and additional conservation measures for more endangered species are not envisaged. The very inclusion of species in the RBRF is one of the legal formalizing signs of rare or endangered species (subspecies, populations), as species of legal protection, formally separated from all other representatives of the animal and plant world.

Another issue was identified in Article 60 of the Federal Law "On environmental protection" which is very declarative and explicitly only prohibits any economic use of species listed in the Red Books and any activity that leads to a reduction in their numbers and worsens their habitat. However, laws themselves make exceptions to these cases and allow the harvesting of species listed in RBRF in order to conserve ABR; monitoring the state of their populations; their artificial reproduction or acclimatization; conducting a traditional way of life and traditional economic activities of indigenous peoples. Sometimes this may hide the economic use of RES that does not comply with the law, which requires a more precise specification of such exceptional cases not only in relation to the extraction but also the circulation of rare species.

The third issue is related to the procedures for the protection of endangered and threatened species. Up to date, part 2 of article 60 of the Federal law "On environmental protection" provides for the protection of rare and endangered plants, animals, and other organisms and for preserving their genetic fund in cryobanks and in artificially created habitats, but the procedure for these processes has not yet been developed and approved, which reduces the effectiveness of the protection of species listed in the RBRF.

Finally, the review of key legislation relevant to the protection of endangered and threatened species in the Russian Federation shows that climate change was not considered in the drafting of the legislation. Despite the huge increase in knowledge of climate change impacts in recent decades, legislation is still limited in how it takes these impacts into account when it comes to protecting endangered and threatened species. There is scope, however, to account for climate change impacts on the endangered and threatened species provided in the Global Target 10 of the strategy and action plan for the conservation of biological diversity of the Russian Federation. In order for policymakers to be able to consider climate change in developing new legislation, or in amending or implementing current legislation, there needs to be an effective information flow between the scientific community and policymakers.

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References

1. Barnosky, A.D.; Matzke, N.; Tomiya, S.; Wogan, G.O.; Swartz, B.; Quental, T.B.; Ferrer, E.A. Has the Earth's sixth mass extinction already arrived? *Nature* **2011**, *471*, 51–57. [CrossRef] [PubMed]
2. Pimm, S.L.; Jenkins, C.N.; Abell, R.; Brooks, T.M.; Gittleman, J.L.; Joppa, L.N.; Sexton, J.O. The biodiversity of species and their rates of extinction, distribution, and protection. *Science* **2014**, *344*, 1246752. [CrossRef]
3. Halpern, B.S.; Frazier, M.; Potapenko, J.; Casey, K.S.; Koenig, K.; Longo, C.; Walbridge, S. Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nat. Commun.* **2015**, *6*, 7615. [CrossRef] [PubMed]
4. McCauley, D.J.; Pinsky, M.L.; Palumbi, S.R.; Estes, J.A.; Joyce, F.H.; Warner, R.R. Marine defaunation: Animal loss in the global ocean. *Science* **2015**, *347*, 1255641. [CrossRef]
5. Islam, M.S.; Tanaka, M. Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: A review and synthesis. *Mar. Pollut. Bull.* **2004**, *48*, 624–649. [CrossRef]
6. Lotze, H.K.; Lenihan, H.S.; Bourque, B.J.; Bradbury, R.H.; Cooke, R.G.; Kay, M.C.; Jackson, J.B. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* **2006**, *312*, 1806–1809. [CrossRef]
7. Pauly, D.; Watson, R.; Alder, J. Global trends in world fisheries: Impacts on marine ecosystems and food security. *Philos. Trans. R. Soc. B Biol. Sci.* **2005**, *360*, 5–12. [CrossRef]
8. Hoegh-Guldberg, O. The impact of climate change on coral reef ecosystems. In *Coral Reefs: An Ecosystem in Transition*; Springer: Dordrecht, The Netherlands, 2011; pp. 391–403.
9. Poloczanska, E.S.; Brown, C.J.; Sydeman, W.J.; Kiessling, W.; Schoeman, D.S.; Moore, P.J.; Richardson, A.J. Global imprint of climate change on marine life. *Nat. Clim. Chang.* **2013**, *3*, 919–925. [CrossRef]
10. Carpenter, K.E.; Abrar, M.; Aeby, G.; Aronson, R.B.; Banks, S.; Bruckner, A.; Wood, E. One-third of reef-building corals face elevated extinction risk from climate change and local impacts. *Science* **2008**, *321*, 560–563. [CrossRef]
11. Dulvy, N.K.; Fowler, S.L.; Musick, J.A.; Cavanagh, R.D.; Kyne, P.M.; Harrison, L.R.; White, W.T. Extinction risk and conservation of the world's sharks and rays. *eLife* **2014**, *3*, e00590. [CrossRef]
12. Polidoro, B.A.; Carpenter, K.E.; Collins, L.; Duke, N.C.; Ellison, A.M.; Ellison, J.C.; Hong Yong, J.W. The loss of species: Mangrove extinction risk and failure of critical exosystem services. *PLoS ONE* **2010**, *5*, e10095. [CrossRef]
13. Short, F.T.; Polidoro, B.; Livingstone, S.R.; Carpenter, K.E.; Bandeira, S.; Bujang, J.S.; Zieman, J.C. Extinction risk assessment of the world's seagrass species. *Biol. Conserv.* **2011**, *144*, 1961–1971. [CrossRef]
14. Collette, B.B.; Carpenter, K.E.; Polidoro, B.A.; Juan-Jordá, M.J.; Boustany, A.; Die, D.J.; Yáñez, E. High value and long life—Double jeopardy for tunas and billfishes. *Science* **2011**, *333*, 291–292. [CrossRef] [PubMed]
15. Salafsky, N.; Salzer, D.; Stattersfield, A.J.; Hilton-Taylor, C.R.A.I.G.; Neugarten, R.; Butchart, S.H.; Wilkie, D. A standard lexicon for biodiversity conservation: Unified classifications of threats and actions. *Conserv. Biol.* **2008**, *22*, 897–911. [CrossRef] [PubMed]
16. Lotze, H.K.; Coll, M.; Magera, A.M.; Ward-Paige, C.; Airoldi, L. Recovery of marine animal populations and ecosystems. *Trends Ecol. Evol.* **2011**, *26*, 595–605. [CrossRef]
17. Akçakaya, H.R.; Bennett, E.L.; Brooks, T.M.; Grace, M.K.; Heath, A.; Hedges, S.; Young, R.P. Quantifying species recovery and conservation success to develop an IUCN Green List of Species. *Conserv. Biol.* **2018**, *32*, 1128–1138. [CrossRef]
18. Chapron, G.; Epstein, Y.; Trouwborst, A.; López-Bao, J.V. Bolster legal boundaries to stay within planetary boundaries. *Nat. Ecol. Evol.* **2017**, *1*, 86. [CrossRef]
19. Gibbs, K.E.; Currie, D.J. Protecting endangered species: Do the main legislative tools work? *PLoS ONE* **2012**, *7*, e35730. [CrossRef]
20. Hoekstra, J.M.; Fagan, W.F.; Bradley, J.E. A critical role for critical habitat in the recovery planning process? Not yet. *Ecol. Appl.* **2002**, *12*, 701–707. [CrossRef]
21. Langpap, C.; Kerkvliet, J. Allocating conservation resources under the Endangered Species Act. *Am. J. Agric. Econ.* **2010**, *92*, 110–124. [CrossRef]
22. Neel, M.C.; Leidner, A.K.; Haines, A.; Goble, D.D.; Scott, J.M. By the numbers: How is recovery defined by the Endangered Species Act? *Bioscience* **2012**, *62*, 646–657. [CrossRef]
23. Scott, J.M.; Goble, D.D.; Svancara, L.K.; Pidgorna, A. By the numbers. In *The Endangered Species Act at Thirty*; Goble, D.D., Scott, J.M., Davis, F.W., Eds.; Island Press: Washington, DC, USA, 2006; pp. 16–35.

24. Suckling, K.F.; Taylor, M. Critical habitat and recovery. In *The Endangered Species Act at Thirty*; Goble, D.D., Scott, J.M., Davis, F.W., Eds.; Island Press: Washington, DC, USA, 2006; pp. 75–89.
25. Greenwald, D.N.; Suckling, K.F.; Taylor, M. The listing record. In *The Endangered Species Act at Thirty: Conserving Biodiversity in Human-Dominated Landscapes*; Scott, J.M., Goble, D.D., Davis, F.W., Eds.; Island Press: Washington, DC, USA, 2006; pp. 51–67.
26. Wilcove, D.S.; McMillan, M.; Winston, K.C. What exactly is an endangered species—An analysis of the United States endangered species list—1985–1991. *Conserv. Biol* **1993**, *7*, 87–93. [CrossRef]
27. Wilcove, D.S.; Master, L.L. How many endangered species are there in the United States? *Front. Ecol. Environ.* **2005**, *3*, 414–420. [CrossRef]
28. Avkhadeev, V.R. Legal Regulation of Arctic Marine Bioresources Development: International Law and State Cooperation Mechanisms. *Pravo. Zhurnal Vyss. Shkoly Ekon.* **2020**, *4*, 216–239. [CrossRef]
29. Avkhadeev, V.R. Fishing of Marine Living Resources in the Arctic States: Legal Problems. *J. Foreign Legis. Comp. Law* **2021**, *17*, 104–116. [CrossRef]
30. Bekyashev, D.K. Issues of protection of marine mammals in the legislation of a number of states. *Rybn. Hozyajstvo* **2021**, *5*, 33–42.
31. Kasprova, Y.A.; Kudelkin, N.S. Legal regulation of fishing out cetaceans and holding them in captivity for the cultural and educational purposes: The experience of foreign countries. *Ecol. Parvo* **2019**, *4*, 30–35.
32. Karpukhin, M.Y. Criminal Law Protection of Objects of the Animal World. *Ross. Pravo Obraz. Prakt. Nauka* **2020**, *5*, 76–81. [CrossRef]
33. Ustyantseva, O.V. Legislative regulation of the use and protection of marine mammals in Russia. *Curr. Issues State Law* **2019**, *3*, 515–529. [CrossRef]
34. Government of the Russian Federation. Federal Law No. 52-FZ of 24 April 241995 "On the Animal World". 1995. Available online: https://www.consultant.ru/document/cons_doc_LAW_6542/ (accessed on 19 September 2022).
35. Red Book of the Russian Federation. *Animals*, 2nd ed.; FGBU «VNII Ecologiya»: Moscow, Russia, 2021; 1128p, ISBN 978-5-6047425-0-1.
36. Government of the Russian Federation. Federal Law "On Specially Protected Natural Territories". 1995. Available online: https://www.consultant.ru/document/cons_doc_LAW_6072/ (accessed on 19 September 2022).
37. Government of the Russian Federation. Federal Law "On Environmental Protection" Dated 1 October 2002 No. 7-FZ. 2001. Available online: https://www.consultant.ru/document/cons_doc_LAW_34823/ (accessed on 19 September 2022).
38. Government of the Russian Federation. Federal Law of the Russian Federation No. 166-FZ of 20 December 2004 "On Fishing and the Conservation of Aquatic Biological Resources". 2004. Available online: https://www.consultant.ru/document/cons_doc_LAW_50799/ (accessed on 19 September 2022).
39. Decree of the President of the Russian Federation. N 176 "On the Strategy for Environmental Security of the Russian Federation for the Period up to 2025". 2017. Available online: https://www.prlib.ru/item/681511 (accessed on 19 September 2022).
40. UNEP/CBD/COP/DEC/X/2. Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity. 2010, XX Meeting, Nagoya, Japan. Available online: https://www.cbd.int/doc/decisions/COP-10/cop-10-dec-02-en.pdf (accessed on 19 September 2022).
41. Strategy and Action Plan for the Conservation of Biological Diversity of the Russian Federation. 2014. Available online: https://www.cbd.int/doc/world/ru/ru-nbsap-v2-ru.pdf (accessed on 19 September 2022).
42. Government of the Russian Federation. Approval of the Strategy for the Conservation of Rare and Endangered Species of Animals, Plants and Fungi in the Russian Federation for the Period up to 2030. 2014. Available online: https://www.consultant.ru/document/cons_doc_LAW_159411/efc7d3560f71426e044ea3cd2c0e34bb75ae75c8/ (accessed on 19 September 2022).
43. Ministry of Natural Resources and Environment of the Russian Federation. Order of the Ministry of Natural Resources of Russia dated 05/23/2016 N 306 (as amended on 07/05/2021) "On approval of the Procedure for maintaining the Red Book of the Russian Federation". 2016. Available online: https://www.consultant.ru/document/cons_doc_LAW_202695/ (accessed on 19 September 2022).
44. Government of the Russian Federation. Code of Administrative Offenses of the Russian Federation. 2001. Available online: https://www.consultant.ru/document/cons_doc_LAW_34661/ (accessed on 19 September 2022).
45. Government of the Russian Federation. Criminal Code of the Russian Federation. 1996. Available online: https://www.consultant.ru/document/cons_doc_LAW_10699/ (accessed on 19 September 2022).
46. Government of the Russian Federation. Decree of the Government of the Russian Federation of 24 December 2008 N 1017 "On the Extraction (Catch) of Rare and Endangered Species of Aquatic Biological Resources". 2008. Available online: https://www.consultant.ru/document/cons_doc_LAW_83216/ (accessed on 19 September 2022).
47. Romanova, A.V.; Kustikova, M.A. Comparison of Red Books and Lists in force in the Arctic. *Tzifrovaya Nauka* **2021**, *4*, 4–11.
48. Kirillova, Y.A. Analysis of the Effectiveness of Red Books and Red Lists on the Territory of St. Petersburg and the Leningrad Region. Master's Thesis, St. Petersburg State University, St Petersburg, Russia, 2017; 85p.
49. Konovalov, A.F.; Borisov, M.Y.; Bolotova, N.L. Distribution of rare and vulnerable species of fish and cyclostomes in water bodies of the Vologda Oblast. *Fish. Issues* **2014**, *15*, 21–35.
50. Tkachenko, P.V. Dynamics of the state of populations of rare fish species in the Tendra and Yagorlytsky bays of the adjacent water areas of the Black Sea from 2006 to 2011. *Nat. Alm.* **2012**, *18*, 194–198.
51. Novoselov, A.P. Rare fish species in the regional Red Books of the European North-East of Russia. *Eurasian Union Sci.* **2015**, *19*, 45–48.

52. Novoselov, A.P. On rare fish species in freshwater reservoirs of the Nenets Autonomous Okrug. In Proceedings of the Current Issues of Biodiversity Conservation in the Regions of Russian Federation. Red Book as an Object of State Environmental Expertise, Perm, Russia, 27–29 October 2015; pp. 142–145.
53. Knizhin, I.B.; Bogdanov, B.E.; Kucheryavy, A.V.; Loshakova, Y.V. On the need for changes in the list of pisciformes and fish included in the Red Data Book of the Irkutsk region. *Baikal Zool. J.* **2012**, *3*, 35–38.
54. Shashulovsky, V.A.; Shlyakhtin, G.V.; Ermokhin, M.V.; Malinina, Y.A. Rare and endangered species of lampreys and fish recommended for inclusion in the third edition of the Red Book of the Saratov Region. News of the Saratov University. New episode. Series Chemistry. *Biol. Ecol.* **2016**, *16*, 318–320.
55. Sereda, M.S. The contribution of the Taganay National Park to the study of rare animal species included in the Red Book of the Chelyabinsk Region. Proceedings of the Mordovian State Natural Reserve named after V.I. *PG Smidovich* **2016**, *17*, 195–203.
56. Dolganov, S.M.; Tyurin, A.N. Far Eastern Marine Biosphere Reserve (Russia). *Biodivers. Environ. Far East Reserves* **2014**, *2*, 76–87.
57. Zhevlakov, E.N. Problems of application of the rule on liability for illegal extraction and trafficking of especially valuable wild animals and aquatic biological resources belonging to species listed in the Red Book of the Russian Federation and (or) protected by international treaties of the Russian Federation. *Crim. Law* **2014**, *1*, 26–31.

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Article

Application of the Method Evaluation and Risk Assessment Tool for a Small-Scale Grouper Fishery in Indonesia

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Abstract: Management strategy evaluation using the Method Evaluation and Risk Assessment (MERA) platform was used to evaluate management procedures (MPs) for improving the management of the leopard coral grouper (*Plectropomus leopardus*) fishery in Saleh Bay, Indonesia. This grouper is a valuable species currently under high fishing pressure. It is targeted by small-scale fisheries using a wide range of fishing methods; hence, management recommendations are needed to ensure sustainability. A suite of MPs for data-limited conditions were evaluated for their ability to achieve limit and target biomass reference points (B/B_{MSY} = 0.5 and B/B_{MSY} = 1, respectively), while maintaining a target yield of at least 0.5 MSY. The simulation results suggest that the currently implemented harvest control rules (HCRs) in Saleh Bay (size limit and spatial closure) may not be effective in achieving the management objective to attain the target biomass reference point due to relatively low compliance with the size limit regulation (320 mm total length) and the very small proportion of existing MPA no-take areas (~2.2%). This study recommends that the fisheries management authority explores the feasibility of implementing the total allowable catch (TAC) and seasonal closure in addition to the existing fishing regulations for *P. leopardus* in Saleh Bay.

Keywords: data-limited fishery; management procedure; harvest control rule; size limit; total allowable catch; total allowable effort; spatial closure; management strategy evaluation

Key Contribution: This study is the first application of the Method Evaluation and Risk Assessment tool to inform the small-scale grouper fishery in Indonesia. This is a case study that can serve as a model for other small-scale fisheries in Indonesia and other regions of the world.

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1. Introduction

Grouper (Serranidae) are a key fishery resource with high economic value, and constitute an essential part of the livelihoods of local communities and help provide food security worldwide [1]. Due to increasing demand for these high-value species, total landings of grouper have gradually increased globally. For example, in the 1950s, total landings were around only 50,000 mt, but between 2006 and 2016, they increased from 237,000 mt to almost 450,000 mt annually [2,3]. China (128,000 mt) and Indonesia (100,000 mt) have the highest grouper landings, together accounting for over 60% of the landings reported to FAO [3]. The average annual grouper production in Indonesia was 105,569 mt from 2012 to

2021, representing 27.3% of the annual global production [4]. These catches contributed ~10% to the total landings of demersal and reef-associated fish species and are mainly targeted by small-scale fisheries [1,5].

Groupers are large, long-lived, late maturing, aggregating species, with high economic value, making them highly vulnerable to fisheries pressure and overfishing [3]. A recent assessment reported that 19 of 71 assessed grouper species were in the "threatened" category under the International Union for Conservation of Nature's (IUCN) Red List of Threatened Species [3]. A study of 716 grouper and snapper fisheries globally suggested that about half of them are in overexploited status [1]. These findings were also confirmed by several studies at the country level. For example, a study on the stock status of 12 reef fish species in Palau [6] showed that the high-value species, such as leopard coral grouper (*Plectropomus leopardus*) and squaretail coral grouper (*Plectropomus areolatus*), are prone to overfishing, as demonstrated by their extremely low estimated spawning potential ratio (SPR) of 0.01 and 0.05, respectively, which is well below the generally accepted limit reference point of 0.20. In addition, a study by Mudjirahayu et al. in Cendrawasih Bay, Indonesia [7], suggested that *Plectropomus maculatus* and *Plectropomus oligacanthus* suffered from overfishing, as indicated by larger fishing mortality compared to their natural mortality rate (i.e. $F/M > 1$).

The leopard coral grouper has a wide distribution, mainly in the Western Pacific from southern Japan to Australia (Queensland and Western Australia), and from Thailand, Malaysia, Indonesia, eastward to the Solomon Islands, the Caroline Islands, and Fiji [8–10]. In Indonesia, this species is distributed throughout the country, living mainly in coral reefs at a wide range of depths from 3 to 100 m [11]. Typically, members of the Epinephelidae are slow growing and have low natural mortality rates and a long-life span. However, the coral groupers (*Plectropomus* spp.) have relatively shorter life spans, grow faster, and have higher rates of natural mortality, and so are considered less vulnerable to fishing pressure than other longer-lived grouper species [8]. However, populations of coral groupers continue to decline due to high exploitation and habitat degradation [6,8]. Despite grouper stocks continuing to decline worldwide, management efforts are almost nonexistent in most areas where fishing pressure on grouper populations is high [1,3,8].

Grouper is the main fisheries resource for the West Nusa Tenggara (WNT) province, Indonesia, with annual grouper production contributing 2.9% to the total WNT annual capture fisheries production from 2010 to 2021 [12]. Saleh Bay in Sumbawa is one of the main fishing grounds for grouper fisheries and other WNT fisheries, with an area of ~2087 km^2 [13] (Figure 1). Local fishers in Saleh Bay target leopard coral grouper, spotted coral grouper (*P. maculatus*), orange-spotted grouper (*Epinephelus coioides*), and malabar blood snapper *(Lutjanus malabaricus*) using various fishing gear including bottom longlines, spearguns, and handlines [5]. Length-based assessments of the 12 most targeted grouper and snapper species in Saleh Bay indicated very high fishing mortality and low spawning potential ratio (SPR), suggesting that these species were under high fishing pressure that led to overfishing or even recruitment failure and collapse [5,14].

Management efforts for grouper fisheries in Saleh Bay have been implemented since 2018, through the enactment of Governor Regulation No. 32/2018 for managing 12 highly targeted grouper and snapper species. This regulation stipulates several management measures: catch size limit, fishing gear restriction, spatial closures through improving the management effectiveness of existing marine protected areas, and strengthening enforcement to stop destructive fishing practices (i.e., cyanide and blast fishing) [15]. Nevertheless, the annual fisheries monitoring and evaluation showed that the stock condition of some high-value species (including *P. leopardus*) remained relatively low in 2021, indicated by SPR values < 0.3 [16]. In addition, a study by Efendi et al. [14] suggested that *P. leopardus* in Saleh Bay has a poor conservation status as indicated by the length-at-first capture being shorter than the length-at-maturity (i.e. $L_c/L_m < 1$), and that the yield is lower than optimum as indicated by the ratio of average individual length to the optimum length (L_{mean}/L_{opt}) < 0.9). This prompted the WNT government to prepare a rebuilding stock strategy using a Management Strategy Evaluation (MSE) method for this species if the SPR is below the limit reference point (SPR < 0.2). This approach is also part of the requirements for the Marine Stewardship Council (MSC) certification of the grouper fishery in Saleh Bay [17].

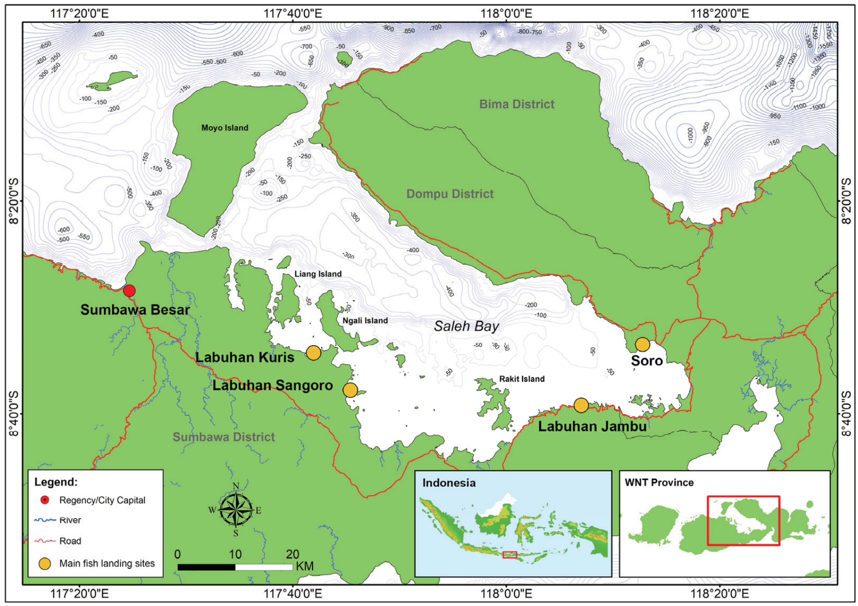

Figure 1. Map of Saleh Bay and four main fish landing monitoring (FLM) sites (yellow circles) around the bay in Sumbawa, West Nusa Tenggara (WNT), Indonesia. Basemap source: Indonesian Geospatial Information Agency (BIG) and the Wildlife Conservation Society (WCS) Indonesia Program. Red boxes on insets show location of Saleh Bay in Indonesia and WNT province.

Management strategy evaluation is an approach that compares the relative efficacy for accomplishing management objectives of various combinations of data gathering systems, techniques of analysis, and subsequent procedures leading to management actions (e.g., harvest strategy) using simulations with a mathematical model of the fishery system [18]. The MSE approach can help fisheries managers to select from a list of candidates, the management strategy that is most likely to achieve management goals or assess the effectiveness of an existing management approach, even when data for conventional stock assessment are not adequate [18,19]. Over the last decade, MSE has become increasingly used for planning, evaluating, and implementing fisheries management plans for data-limited fisheries, frequently including participatory modeling [20]. For example, MSE using the data-limited method toolkit (DLMtool) has been applied to Pacific groundfish species (Pleuronectidae and Scorpaenidae; [21]) in Canada; developing fisheries management plans for barred sand bass (Serranidae) in California, and halibut, red sea urchin, and warty sea cucumber in California [22]; and seven data-limited fisheries in Indonesia [23], including leopard coral grouper [11].

One methodology developed for applying MSE to fisheries is the Method Evaluation and Risk Assessment (MERA) application, which was developed as a tool to evaluate alternative management strategies and aid selection of the approach that is most likely to achieve the desired management objectives [24]. Since MERA applies a quantitative approach and documents all steps in the process, it has advantages over more qualitative approaches to MSE, such as the Productivity Susceptibility Analysis (PSA) [25]. Although often considered a useful tool for management evaluation, a qualitative approach is often subjective and not reproducible [25]. On the other hand, MERA enables clear quantitative documentation of the simulation and analysis process, including the data and assumptions being used, to allow the process to be repeatable [24,25].

This study is part of the process of providing fisheries management advice for *P. leopardus*, the main high-value grouper species targeted by small-scale fisheries in Saleh Bay [26]. The MERA application was applied to evaluate the performance of a suite of different management procedures for achieving fisheries management objectives, particularly the biological objective of achieving sustainable fish stocks and the fishery objective of sustaining yields from the fishery. This study conducted simulations and analysis using MERA to (1) evaluate the performances of 20 management procedures (MPs) to achieve management objectives of *P. leopardus* under current fishery conditions; (2) evaluate the performances of 20 MPs under a stock rebuilding scenario, i.e., under depleted *P. leopardus* stock conditions (biomass below the limit reference point); and (3) evaluate a suite of customized management procedures (19 MPs based on variations of catch reductions, size limits, and length of closure to fishing) as a basis for recommending technical harvest control rules that may be applicable for *P. leopardus* management in Saleh Bay. MERA was also used to evaluate the greatest sources of uncertainty affecting the simulation results of each MP. The results from this study will inform fisheries managers and stakeholders of the management procedures that are simulated to have a high probability of achieving management objectives, such as maintaining biomass within the population and yield from the fishery.

2. Materials and Methods

2.1. Study Area and Fishery

Saleh Bay is situated on Sumbawa Island and is one of the main fishing grounds for ~5800 pelagic, demersal, and reef fishers in WNT Province, Indonesia [5,13,14]. It is a semi-enclosed area with a total area of 2087 km^2 and maximum depth of 324 m [13,27]. Its 128 km long coastline is inhabited by ~67,000 people distributed in 26 coastal villages [13,27]. Saleh Bay encompasses a variety of habitats, including small islands and varied coastal ecosystems such as coral reefs, seagrass, and mangroves, which provide vital habitats for a variety of fish resources [13].

Since 2016, the small-scale grouper and snapper fisheries in Saleh Bay have been intensively studied through the fish landing monitoring (FLM) program by the WNT provincial government, supported by various stakeholders. The FLM program has made it possible to improve the fisheries data quality of Saleh Bay, which also includes data on species, length composition, fishing effort, fishing unit characteristics, and socio-economic data. The FLM program in Saleh Bay focused on four main landing sites (Figure 1) that were agreed upon by the government and stakeholders as monitoring sites, where >80% of the fishing population in Saleh Bay is concentrated [13,15]. The resultant data allowed a fisheries action plan to be formulated through Governor Regulation No. 32/2018, which has been enforced and monitored since September 2018 [15]. This regulation governs the fishing activities targeting 12 main grouper and snapper species in Saleh Bay through (1) limiting the catch size, (2) limiting the size of fish to trade, (3) regulating the specifications of several fishing gears (e.g., hook and mesh size), and (4) recommending fishers and fisher groups to develop an agreement on restricted fishing times to reduce fishing pressures. Most of the data and information used in this study were taken from analyses of data from the FLM program.

Based on the technical and operational characteristics of fishing units, the grouper fishery in Saleh Bay is a small-scale complex fisheries system. This employs multiple types of fishing gear and methods with various boat characteristics and targets a wide range of species. Over 75 grouper and snapper species are caught in Saleh Bay, where *P. leopardus* is among the most targeted species. Among seven fishing gear/methods operating in Saleh Bay, the bottom longline, speargun, and handline contributed about 90% to the grouper catch in Saleh Bay [5], and *P. leopardus* is a high-value species for local and export markets, mainly Hong Kong and Taiwan [28].

Plectropomus leopardus is a protogynous hermaphroditic species, aggregating during spawning to form groups of several hundred fish [10]. In addition, this species also undergoes diandric protogyny, where male *P. leopardus* may develop from either immature or mature females [8]. In eastern Indonesia, spawning of *P. leopardus* occurs from October to January, with reproductive peaks in November and December [29]. The spawning season in higher latitude waters is from May to July in the Okinawa Islands, Japan [30], and from September to December in the Great Barrier Reef, Australia [8].

Species in the *Plectropomus* genus have a high level of inter-specific variation in mean size and age at maturity, with ranges of 35–62 cm TL (mean size), 1.8–4.6 years (age at maturity), and sex change at 42.0–87.4 cm TL [8]. The growth parameters of *P. leopardus* in Saleh Bay (for both sexes combined) used for this analysis were: L_∞ = 719.4 mm, k (year^{-1}) = 0.12, t_0 = −1.17 year, and natural mortality M = 0.16 year^{-1} (n samples = 1159) (Table 1; [26]). The growth parameters for *P. leopardus* vary widely among different regions, e.g., the estimated L_∞ ranges from 416 mm in Cendrawasih Bay (Papua) to 924 mm in South Sulawesi, and k ranges from 0.21 in Southeast Sulawesi to 1.2 year^{-1} in Papua [11]; see also [5,8].

2.2. MERA Operation

MERA uses two inputs: (1) responses from an essential quantitative questionnaire and (2) an optional input of fisheries data in a standardized format. The questionnaire consists of 30 questions which are grouped into three categories: fisheries characteristics (including biological attributes of the target species, 19 questions); suitable fisheries management type (7 questions); and the quality of data (4 questions) [24]. By default, MERA runs its simulation based on the data and information input into the questionnaire. When provided, the standardized fishery data is used for (1) conditioning an operating model when simulating Management Planning, Management Performance, and Risk Assessment modes using closed-loop simulation, and/or (2) estimating the status of exploited stocks in Status Determination Mode [24]. Seven categories of fishery data can be provided: biological parameters, selectivity parameters, historical catch and effort data, catch-at-age data, catch-at-length data, and reference points or other metrics [24].

MERA has three primary modes of operating for providing information on management options or procedures: (1) Management Planning: which entails determining an appropriate management mode; (2) Management Performance: which assesses current management practices; and (3) Calculating current stock status [24]. This study uses the Management Planning mode to evaluate the performance of the management procedures that will inform management. This mode provides a closed-loop simulation testing of numerous management procedures and diagnostics to help managers identify research priorities. The inputs required for MERA analysis are (1) responses from an essential quantitative questionnaire, (2) an optional input of fisheries data in a standardized format, and (3) a selection of management procedures (Figure 2). This study evaluates 20 management procedures conditioned to the current fishery state, based on data and information provided in the questionnaire and fishery data, and in the rebuilding scenario where the current biomass is assumed to be between 30 and 50% of B_{MSY} (biomass at maximum sustainable yield; Figure 2). It also evaluates 19 custom management procedures developed specifically for *P. leopardus* in Saleh Bay.

2.2.1. MERA Questionnaire

The MERA questionnaire consists of 30 questions which are grouped into three categories: (1) fisheries characteristics (including biological attributes of the target species, 19 questions); (2) suitable fisheries management type (7 questions); and (3) the quality of data (4 questions; [24]). The questionnaire was completed through a data workshop involving scientists from the Indonesian Research and Innovation Agency (BRIN), the Fisheries Resources Center of Indonesia (FRCI), Mataram University, the Wildlife Conservation Society (WCS) Indonesia Program, the scientific forum for fisheries management of West

Nusa Tenggara (FIP2B), and the West Nusa Tenggara Marine and Fisheries Agency as the fisheries management authority. The questions in the questionnaire were answered using the best available data, scientific literature, and expert judgment and knowledge of the fishery in the study area. The complete data inputs for the MERA questionnaire are provided in Supplementary Material S1.

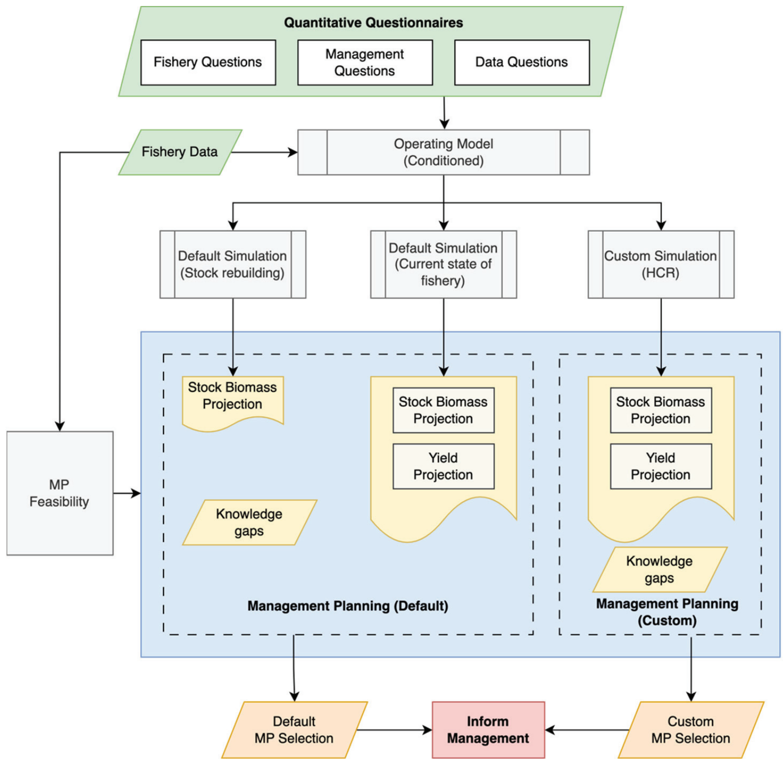

Figure 2. The flow of the MERA simulation performed in this study including data inputs, three modes of closed-loop simulation (default, rebuilding stock, and custom simulations) that result in projected biomass stocks and yields and a yield–biomass trade-off plot, and selection of MPs to inform management of *P. leopardus* in Saleh Bay. Dashed rectangles enclose the default and custom MERA simulations. Adapted with permission from Carruthers et al. [24].

2.2.2. Fishery Data

The available fishery data used in this study consist of (1) biological parameters, (2) selectivity parameters, (3) a time series of historical catch, coefficient of annual variation of catch (cv; estimated using expert judgement at 0.1 per year), and vulnerable abundance index, (4) catch-at-length, and (5) estimated current stock depletion (Table 1; Supplementary Material S2). The biological and selectivity parameters were derived from a study in Saleh Bay [26], a recent fisheries monitoring and evaluation report [16], and studies from other areas (e.g., [31]). The historical catch data were derived from the fisheries statistics data of the WNT Province from 2009 to 2015, and the fish landing monitoring data from 2016 to 2021. We used the catch per unit effort (CPUE) estimation from Efendi et al. [32] as the abundance index for 2009 to 2015, and standardized CPUE estimation from fish landing monitoring data for 2016 to 2021. Catch-at-length data were derived from the fish landing monitoring data from 2016 to 2021. When the fishery data matrix is uploaded, MERA overrides the corresponding data and information in the questionnaire and replaces it with data from the fishery data matrix. MERA then uses the fishery data matrix in conditioning the operating model to the current state of the fishery.

Table 1. Fishery parameters and data used for the Method Evaluation and Risk Assessment (MERA) simulations of different management procedures for *P. leopardus* in Saleh Bay, Indonesia. SPR = spawning potential ratio. FIPP2B = scientific forum for fisheries management of West Nusa Tenggara province.

Parameters	Value	References
Life history and stock		
Maximum age (year)	26	
M (year^{-1})	0.16	
Von Bertalanffy L_{inf} parameter (cm)	71.9	
Von Bertalanffy k parameter	0.12	
Von Bertalanffy t_0 parameter (year)	−1.17	Mathews and Samuels [31];
Length-weight parameter a	0.0182	Agustina et al. [16,26,33]
Length-weight parameter b	2.97	
Length at 50% maturity (cm)	38.8	
Length at 95% maturity (cm)	41.8	
Length at first capture (cm)	34.6	
Stock depletion	0.32	Estimated from current SPR relative to general MSY equilibrium model (Goethel et al. [34]; Hoshino et al. [35])
Catch data		
Range of total annual catch (kg), from 2016 to 2021	3773.1–5629.4	FIP2B fish landing monitoring data and analyses from 2016 to 2021 (unpublished)
Catch-at-length (cm), from 2016 to 2021	Size class range: 20–64 cm; n = 4115	

The stock depletion value for MERA's model conditioning was calculated from the spawning potential ratio (SPR) estimation of *P. leopardus* in Saleh Bay from 2017 to 2022 using the equilibrium model of maximum sustainable yield (MSY) and the SPR relationship from Goethel et al. [34]. Based on the simulated equilibrium MSY model of Goethel et al., the SPR values at MSY range from 0.24 to 0.38, depending on the assumption used for the stock-recruit model. The LBSPR values of *P. leopardus* in Saleh Bay from 2017 to 2022 ranged from 0.12 to 0.32 (mean = 0.24) [16]. Given the unknown stock-recruit relationship for *P. leopardus* in Saleh Bay, we chose Goethel's SPR at MSY value (SPR_{MSY}) = 0.38 as a precautionary approach. Using the estimated SPR at MSY (SPR_{MSY}) at 0.38, we approximated the current biomass of *P. leopardus* in Saleh Bay as 0.63 of B_{MSY} (SPR_{curr} at 0.24 divided by SPR_{MSY} at 0.38). Using the general equilibrium MSY model from Hoshino et al. [35] where $B_{MSY} = 0.5\ B_0$, we estimated the current biomass level relative to unfished biomass (B_{curr}/B_0) or depletion of the *P. leopardus* $\approx 0.32\ B_0$ ($0.63 \times 0.5\ B_0$). Thus, we used the depletion value (D) = 0.32 in the uploaded fisheries data matrix as the reference value for the MERA simulation. The more detailed fishery data matrix used in this study is presented in Supplementary Material S2.

2.2.3. Management Procedures

In this simulation, we used two groups of management procedure (MP): MERA's default MPs and a set of custom MPs. The default MPs are MERA's "Top 20" MPs, a subset of 20 MPs that regularly rank among the top performers across a wide range of operating models. These procedures include those based on the total allowable catch (TAC), total allowable effort (TAE), size limits (SzLim), and spatial closures (Table 2; see Supplementary Material S3 for a detailed description of these 20 MPs). The 20 customized MPs (Table 2) included variations in TAC and TAE where the nominal total allowable catch and effort are reduced by 10 to 25% (with a 5% annual increment) of the current catch and effort level (mean of the five years from 2017 to 2021). We also tested a scenario of seasonal closure of fishing (reducing annual fishing effort), varying from a one-month closure to four months, to mimic the TAE. We also tested a wide range of catch size limits (from 25 to 40 cm in total length [TL]), starting from a hypothetical catch size smaller than the 50% length-at-maturity (L_{50} = 38.8 cm TL, Table 1), where the current agreed size limit for the *P. leopardus* in Saleh Bay is only 32 cm. The size limits tested were 25 cm TL, and then in 2 cm increments from 28 to 40 cm TL. To perform the customized MP simulation, a file containing a series of R-scripts was uploaded to MERA (Supplementary Material S4). The reference of R-scripts for operating model conditioning was accessed from https://openmse.com/ (accessed on 13 November 2022).

Table 2. MERA's default and custom management procedures (MPs) are simulated in this study. Descriptions of the default MPs are from DLM Tool Documentation 6.0.6 by T. Carruthers, Q Huynh, and A. Hordyk (https://dlmtool.openmse.com/reference/index.html; accessed on 2 August 2023). Full descriptions of MPs are given in Supplementary Material S3.

Management Procedures	Procedures Evaluated
MERA's Default MPs (*n* = 20)	
Total Allowable Catch (TAC, 13)	**DBSRA** (Depletion-Based Stock Reduction Analysis) **DBSRA_40** (Depletion-Based Stock Reduction Analysis 40) **DBSRA4010** (Depletion-Based Stock Reduction Analysis 4010) **DCAC** (Depletion Corrected Average Catch) **DCAC_40** (Depletion Corrected Average Catch 40) **DD** (Delay-Difference Stock Assessment) **DD4010** (Delay-Difference Stock Assessment 4010) **MCD** (Mean Catch Depletion) **MCD4010** (Mean Catch Depletion 4010) **Fratio** (F and M ratio) **HDAAC** (Hybrid Depletion Adjusted Average Catch) **IT10** (Iterative Index Target 10%) **IT5** (Iterative Index Target 5%)
Total Allowable Effort (TAE, 3)	**DDe** (Effort-based Delay-Difference Stock Assessment) **DDe75** (Effort-based Delay-Difference Stock Assessment 75%) **ITe10** (Index Target Effort-Based 10%)
Size Limit (2)	**Matlenlim** (Size limit at length-at-maturity) **Matlenlim2** (Size limit at 110% length-at-maturity)
Spatial Closure/Marine Protected Area (2)	**MRnoreal** (Spatial closure—no reallocation of effort) **MRreal** (Spatial closure—with reallocation of effort)
Custom MPs (*n* = 19)	
Total Allowable Catch (TAC; 4)	**Index_10_TAC** (Reduction of 10% from current catch) **Index_15_TAC** (Reduction of 15% from current catch) **Index_20_TAC** (Reduction of 20% from current catch) **Index_25_TAC** (Reduction of 25% from current catch)
Total Allowable Effort (TAE; 4)	**Index_10_Eff** (Reduction of 10% from current effort) **Index_15_Eff** (Reduction of 15% from current effort) **Index_20_Eff** (Reduction of 20% from current effort) **Index_25_Eff** (Reduction of 25% from current effort)
Size Limit (8)	**SL_25** (set catch size limit at 25 cm) **SL_28 to SL_40** (set catch size limit at between 28 to 40 cm, with 2 cm increment)
Seasonal closure (3)	**SC_2** (no fishing for 2 month) **SC_3** (no fishing for 3 month) **SC_4** (no fishing for 4 month)

2.2.4. Evaluating the Performance and Selecting Management Procedures

We evaluated and selected the MPs as management recommendations based on their performances in achieving biomass (B) limit and target reference points relative to biomass at maximum sustainable yield (B_{MSY}) and the probabilities of achieving a reasonably high yield (Y) relative to the current yield (Y_{curr}). We selected both the MERA default and custom MPs as management recommendations when the MPs (1) achieve the $B/B_{MSY} > 1$ target in year 50, (2) achieve the $B/B_{MSY} > 1$ target in medium-term stock rebuilding scenario (year 20), (3) achieve the $B/B_{MSY} > 1$ in long-term stock rebuilding scenario (year 50), and (4) achieve $Y_t/Y_{curr} > 1$ in year 50. MPs not meeting these criteria were filtered out and not selected for further discussion in this study.

2.2.5. Biomass and Yield Projections

Probability projection plots of biomass and yield for each selected MP were also provided to achieve the pre-determined reference points for the current fisheries condition (base simulation) and whether they could rebuild the hypothetical depleted stock (stock rebuilding simulation). The probability projection plots are simulated in two different time frames. The base simulation evaluates biomass and yield projections in 0–10 years (short-term) and 50 years (long-term), while the stock rebuilding simulation plots the projections in 20 years (medium-term) and 46–50 years (hereafter termed "50" years; long-term). In the stock rebuilding simulation, the MP will likely provide a high probability of rebuilding the stock when the median estimates of the probability plot exceed the target reference point $B/B_{MSY} > 1$.

Tradeoff plots of project biomass (*x*-axis) and yield (*y*-axis) for the selected MPs are displayed to identify those MPs predicted to achieve biomass and yield reference points in the long-term. Two yield–biomass plots are provided with different probabilities of biomass targets shown on the *x*-axis: the probability of maintaining biomass level above (a) the limit reference point $B > 0.5B_{MSY}$ and (b) the target reference point $B > B_{MSY}$ over a long-term period ("50" years).

2.2.6. Source of Uncertainties and Variation in Yield Projections

Parameter inputs from answers to the MERA questionnaire and the fishery data matrix and the operational models used in MERA also contribute to uncertainties, causing variation in the probability projection of long-term yield. Uncertainty plots for the selected MPs are provided to identify the source of uncertainties that affect the variability in the projection of the long-term yield (LTY) as a percentage (%LTY).

3. Results

3.1. Selection of Available Management Procedures

Based on the selection criteria, ten default MPs and eight custom MPs were selected as potential management recommendations for *P. leopardus* in Saleh Bay (Table 3). These MPs were selected based on their performance in keeping biomass above the limit reference point (LRP) and simultaneously achieving the biomass and yield target reference points (TRP) in the 50-year simulation. For example, DBSRA4010 (depletion-based stock reduction analysis) had the highest probability (98.7%) of maintaining biomass above the limit reference point ($B > 50\%\ B_{MSY}$), while at the same time being able to achieve the target biomass ($B/B_{MSY} > 1$) and target yield ($Yt/Y_{curr} = 1$). Only simulations for the selected default MPs achieved the limit and target reference points of biomass in the stock rebuilding scenario (Table 3). In addition to the TAC MPs, the selected MPs included those using a seasonal closure from two to four months (i.e., a form of total allowable effort) and size limits from the current size limit of 32 cm total length to 40 cm total length (Table 3). In contrast, no custom MPs performed well under the stock rebuilding scenario. All the default total allowable effort (TAE), size limit, and spatial closure (MPA)-based MPs showed poor performance in achieving the LRP and TRP for biomass (Table 3).

Table 3. Selected default and custom management procedures (MPs) based on the selection criteria for *P. leopardus* in Saleh Bay, West Nusa Tenggara, Indonesia (* are the main considerations in MP selection).

MP	MP Type	MP Group	Simulation for Current Fishery Condition				Simulation for Stock Rebuilding Scenario	
			Mean Prob. Biomass > 50%B_{MSY} (Year 1–10)	Mean Prob. Biomass > B_{MSY} (Year 1–10)	Achieve B/BMSY > 1 in year 46–50 (1 = Yes; 0 = No) *	Achieve Yt/Ycurr = 1 in Year 46–50 (1 = Yes; 0 = No) *	Achieve B/BMSY > 1 in year 1–10 or short-term HCR (1 = Yes; 0 = No)	Achieve B/BMSY > 1 in year 46–50 or long-term HCR (1 = Yes; 0 = No) *
DBSRA4010	TAC	Default	98.7	27.1	1	1	1	1
DD4010	TAC	Default	98.7	28.4	1	1	1	1
MCD4010	TAC	Default	98.7	30.1	1	1	1	1
DBSRA	TAC	Default	98.6	20.3	1	1	1	1
DCAC_40	TAC	Default	98.6	22.2	1	1	1	1
DCAC	TAC	Default	98.6	22.8	1	1	1	1
MCD	TAC	Default	98.6	26.5	1	1	1	1
HDAAC	TAC	Default	98.6	27.1	1	1	1	1
DD	TAC	Default	97.8	11.0	1	1	0	1
DBSRA_40	TAC	Default	97.4	11.4	1	1	0	1
SC_4	TAE	Custom	94.59	3.53	1	1	0	0
SC_3	TAE	Custom	91.36	2.81	1	1	0	0
SC_2	TAE	Custom	89.48	2.4	1	1	0	0
SL_40	SzLim	Custom	91.05	3.22	1	1	0	0
SL_38	SzLim	Custom	88.76	3.01	1	1	0	0
SL_36	SzLim	Custom	88.25	2.81	1	1	0	0
SL_34	SzLim	Custom	86.35	2.7	1	1	0	0
SL_32	SzLim	Custom	85.53	2.2	1	1	0	0

3.2. Yield–Biomass Trade-Offs

The yield–biomass trade-offs plot of the 18 selected MPs showed that only DBSRA and DBSRA4010 (TAC-based MPs; output control) had a high probability (>0.75) of exceeding the biomass limit (B > 50% B_{MSY}) and target (B > B_{MSY}) reference points, while still achieving a reasonably high probability (>0.75) of yield being >50% of the reference yield (i.e., the yield at MSY) over the long term (Figure 3). The custom size limit and seasonal closure MPs showed a high probability (>0.75) of yield being >50% yield at MSY with lower probability (0.5–0.75) in achieving biomass limit and reference points. The other MPs (TAC-based; output control) performed very well in maintaining the biomass condition (probability > 0.75) but had lower probability (<0.75) of having a reasonably high yield (Figure 3).

3.3. Biomass Projections

3.3.1. Biomass Probability Projections

Based on the trade-off plots (Figure 3), 10 MPs were selected as management recommendations for further analysis (Tables 4 and 5). Eight of these ten MP projections (all except for the SL_32 and SL_34) had a very high probability (>90%) of being above the biomass limit reference point (B > 0.5 B_{MSY}) over the first 10 years of simulation (i.e., 2024 to 2033, Table 4). In contrast, all of these MPs, except DBSRA and DBSRA4010, had low probabilities (<0.6) of achieving the biomass target reference point (B > B_{MSY}) (Table 5).

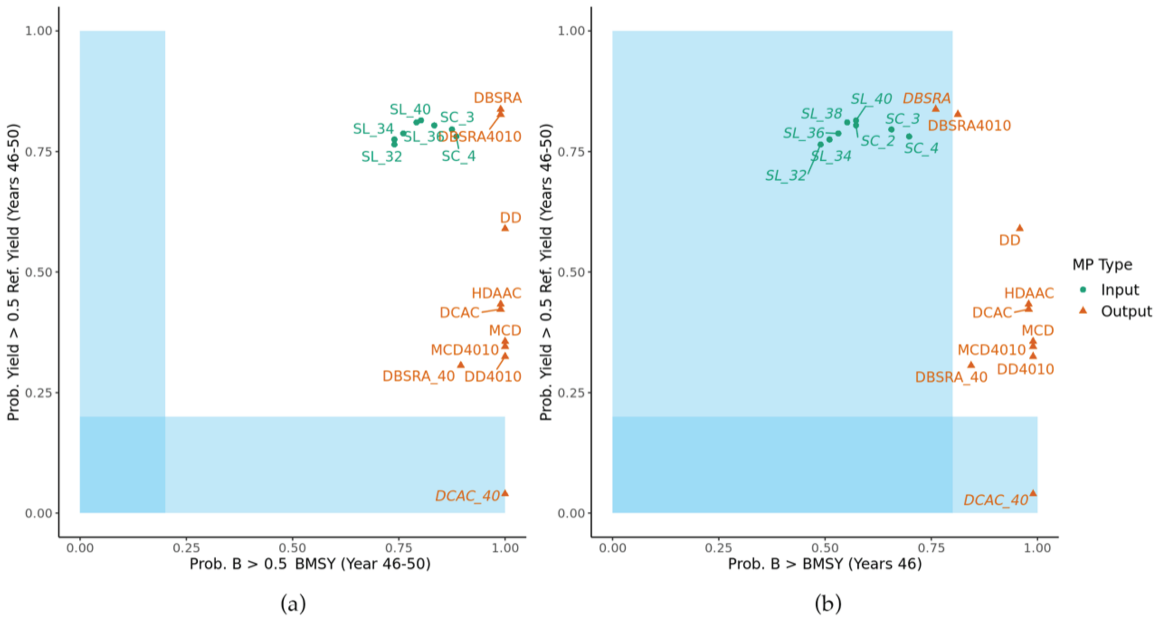

Figure 3. The long-term yield–biomass trade-off of the selected management procedures for *P. leopardus* in Saleh Bay, West Nusa Tenggara showing the probabilities of achieving (**a**) the limit reference biomass (B > $0.5B_{MSY}$) and (**b**) the target reference biomass (B > B_{MSY}). The *y*-axis is the probability of obtaining more than half the reference (Ref.) yield (i.e., 0.5 B_{MSY}). Input MPs (size limits, total allowable effort, or spatial closures) = green lettering; output MPs (total allowable catch) = gold lettering. Blue shades show probability thresholds between 0–0.2 and 0–0.8. The top-right region represents better performance, and the bottom-left represents worse performance. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

Table 4. Probabilities of the 10 selected management procedures (MP) achieving the limit reference point of biomass (B > 0.5 B_{MSY}) for *P. leopardus* in Saleh Bay Indonesia over 10 years of simulation (2024–2033). Probabilities > 90% = green shaded, 50–90% = orange shaded. MP type denotes the class of MP according to the type of advice it provides: Total Allowable Catch (TAC), Total Allowable Effort (TAE), and Size Limit (SzLim). Full descriptions of MPs are given in Supplementary Material S3.

MP	MP Type	MP Group	Year									
			2024	2025	2026	2027	2028	2029	2030	2031	2032	2033
DBSRA	TAC	Default	95.8	97.9	97.9	97.9	100	100	100	100	100	100
DBSRA4010	TAC	Default	95.8	97.9	99	99	100	100	100	100	100	100
SC_2	TAE	Custom	95.8	94.8	94.8	95.8	92.7	92.7	92.7	92.7	92.7	91.7
SC_3	TAE	Custom	95.8	94.8	94.8	95.8	94.8	95.8	95.8	95.8	95.8	94.8
SC_4	TAE	Custom	95.8	95.8	95.8	96.9	95.8	96.9	96.9	97.9	96.9	95.8
SL_32	SzLim	Custom	95.8	94.8	94.8	94.8	93.8	90.6	91.7	89.6	89.6	87.5
SL_34	SzLim	Custom	95.8	94.8	94.8	94.8	93.8	90.6	91.7	89.6	89.6	89.6
SL_36	SzLim	Custom	95.8	94.8	94.8	94.8	93.8	91.7	91.7	91.7	90.6	90.6
SL_38	SzLim	Custom	95.8	94.8	94.8	94.8	93.8	93.8	92.7	92.7	91.7	91.7
SL_40	SzLim	Custom	95.8	94.8	94.8	94.8	94.8	93.8	94.8	93.8	92.7	91.7

Table 5. Probabilities of 10 selected management procedures (MP) achieving the target reference point of biomass (B > B_{MSY}) for *P. leopardus* in Saleh Bay Indonesia over 10 years of simulation (2024–2033). Probabilities > 75% = green shaded, 50–75% = orange shaded, and ≤50% = red shaded. MP type denotes the class of MP according to the type of advice it provides: Total Allowable Catch (TAC), Total Allowable Effort (TAE), and Size Limit (SzLim). Full descriptions of MPs are given in Supplementary Material S3.

MP	MP Type	MP Group	Year									
			2024	2025	2026	2027	2028	2029	2030	2031	2032	2033
DBSRA	TAC	Default	32.3	36.5	46.9	56.2	58.3	66.7	67.7	74	80.2	84.4
DBSRA4010	TAC	Default	32.3	36.5	49	57.3	62.5	68.8	72.9	79.2	86.5	86.5
SC_2	TAE	Custom	32.3	33.3	35.4	36.5	38.5	43.8	46.9	47.9	46.9	44.8
SC_3	TAE	Custom	32.3	33.3	35.4	38.5	42.7	46.9	49	50	52.1	58.3
SC_4	TAE	Custom	32.3	34.4	38.5	42.7	51	54.2	55.2	54.2	60.4	61.5
SL_32	SzLim	Custom	32.3	34.4	33.3	37.5	36.5	38.5	40.6	42.7	43.8	44.8
SL_34	SzLim	Custom	32.3	34.4	34.4	37.5	36.5	40.6	40.6	43.8	43.8	44.8
SL_36	SzLim	Custom	32.3	35.4	36.5	37.5	39.6	43.8	44.8	43.8	45.8	46.9
SL_38	SzLim	Custom	32.3	35.4	37.5	38.5	41.7	44.8	45.8	45.8	46.9	49
SL_40	SzLim	Custom	32.3	35.4	37.5	38.5	42.7	47.9	47.9	45.8	49	47.9

3.3.2. Biomass Projections Plots

Under the current fishery conditions, all the selected MPs performed well in achieving the biomass target reference point ($B/B_{MSY} = 1$) over the long-term simulation of 50 years (Figure 4). However, almost all MPs (except for DBSRA and DBSRA4010) have relatively higher uncertainty as shown by a wider probability interval (blue and light blue shades), and have 50% probability intervals (blue shade) being lower than the biomass target reference point.

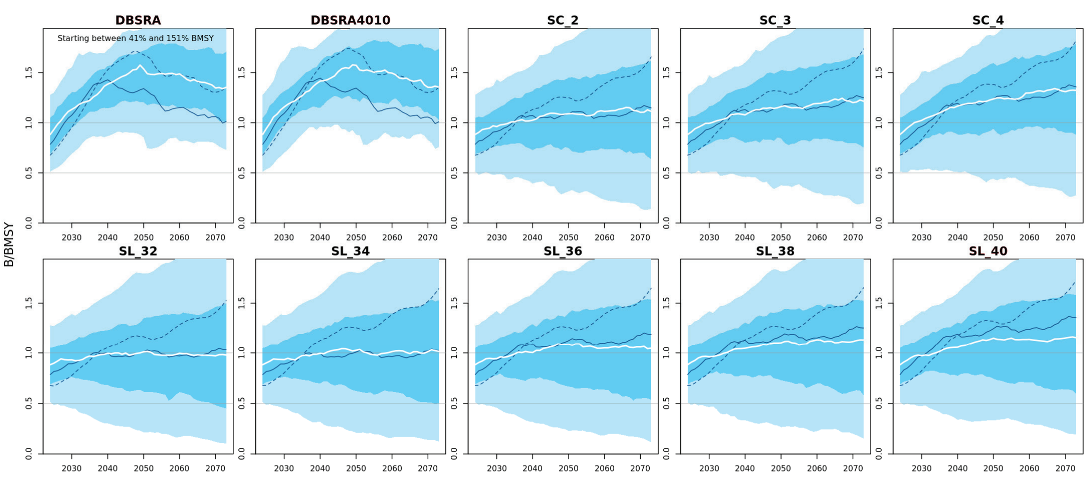

Figure 4. Biomass projection (B/B_{MSY}) relative to the target (B/BMSY = 1) and limit reference points (B/BMSY = 0.5) for *P. leopardus* in Saleh Bay, West Nusa Tenggara, Indonesia. Light blue = 90% probability interval, dark blue = 50% probability interval, white line = median estimate, and two dark blue lines = example simulations. The two horizontal grey lines represent B_{MSY} as the target reference point and 0.5 B_{MSY} as the limit reference point. DBSRA = depletion-based stock reduction analysis; DBSRA4010 = depletion-based stock reduction analysis with 40–10 rule; SC_2 to SC_4 = two to four months seasonal closure; SL_32 to SL-40 = size limit at 32 to 40 cm. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

Under the stock rebuilding scenario where the stock depletion rate is simulated between 30 and 50% B_{MSY}, only DBSRA and DBSRA4010 performed well in achieving the target reference point within 20 years (Figure 5) and 50 years (Figure 6). The other MPs showed poor performance even in maintaining the biomass above the limit reference point in simulations within 20 and 50 years. A no-fishing scenario (NFref) projection is also plotted as a reference for other MPs (Figures 5 and 6). For example, if a no-fishing regulation is implemented, the stock will likely be rebuilt to B/BMSY > 1 in ~5 years in the medium term and 15 years in the long-term simulation. This suggests that only DBSRA and DBSRA4010 will likely perform well in rebuilding the *P. leopardus* stock if its biomass falls below the limit reference point.

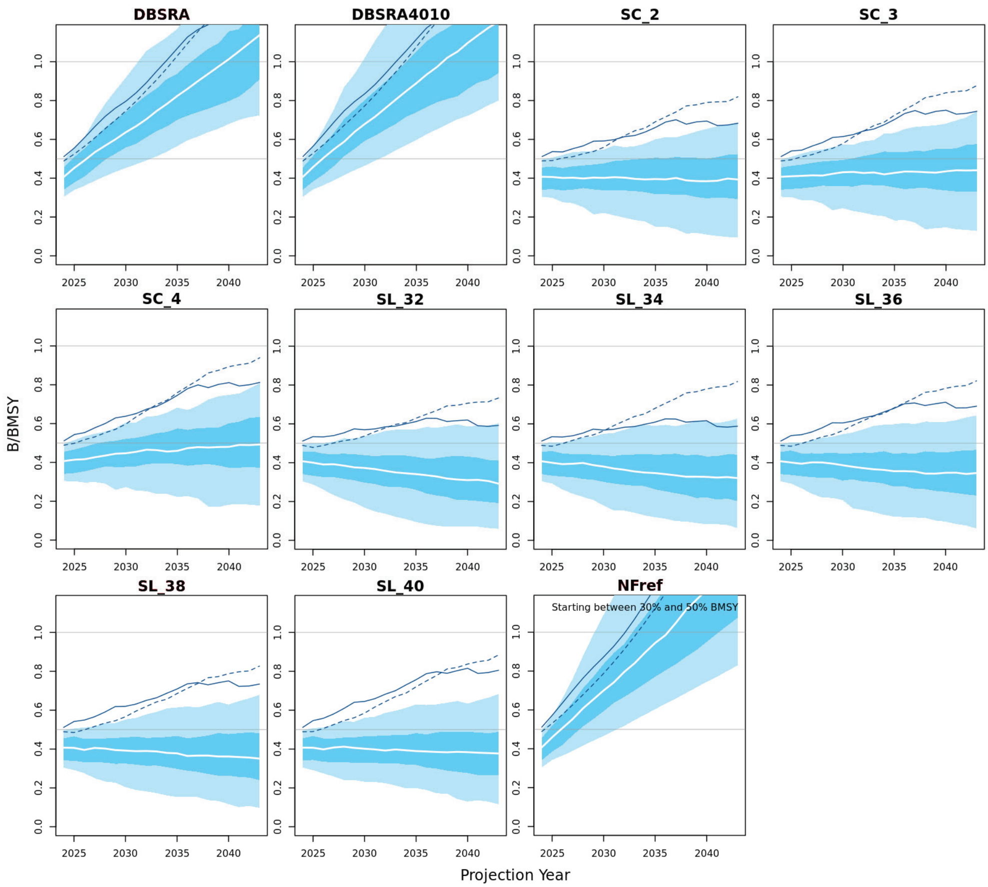

Figure 5. Short-term (20 years) biomass projection under a stock rebuilding scenario (started from the stock condition at 30 to 50% B_{MSY}) relative to the target and limit reference points (B/B_{MSY}) for the *P. leopardus*. The light blue represents a 90% probability interval, the dark blue represents a 50% probability interval, the white line is the median estimate, and the two dark blue lines are example simulations. The two horizontal grey lines represent target and limit reference points. DBSRA = depletion-based stock reduction analysis; DBSRA4010 = depletion-based stock reduction analysis with 40–10 rule; SC_2 to SC_4 = two to four months seasonal closure; SL_32 to SL-40 = size limit at 32 to 40 cm; NFref = no-fishing reference. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

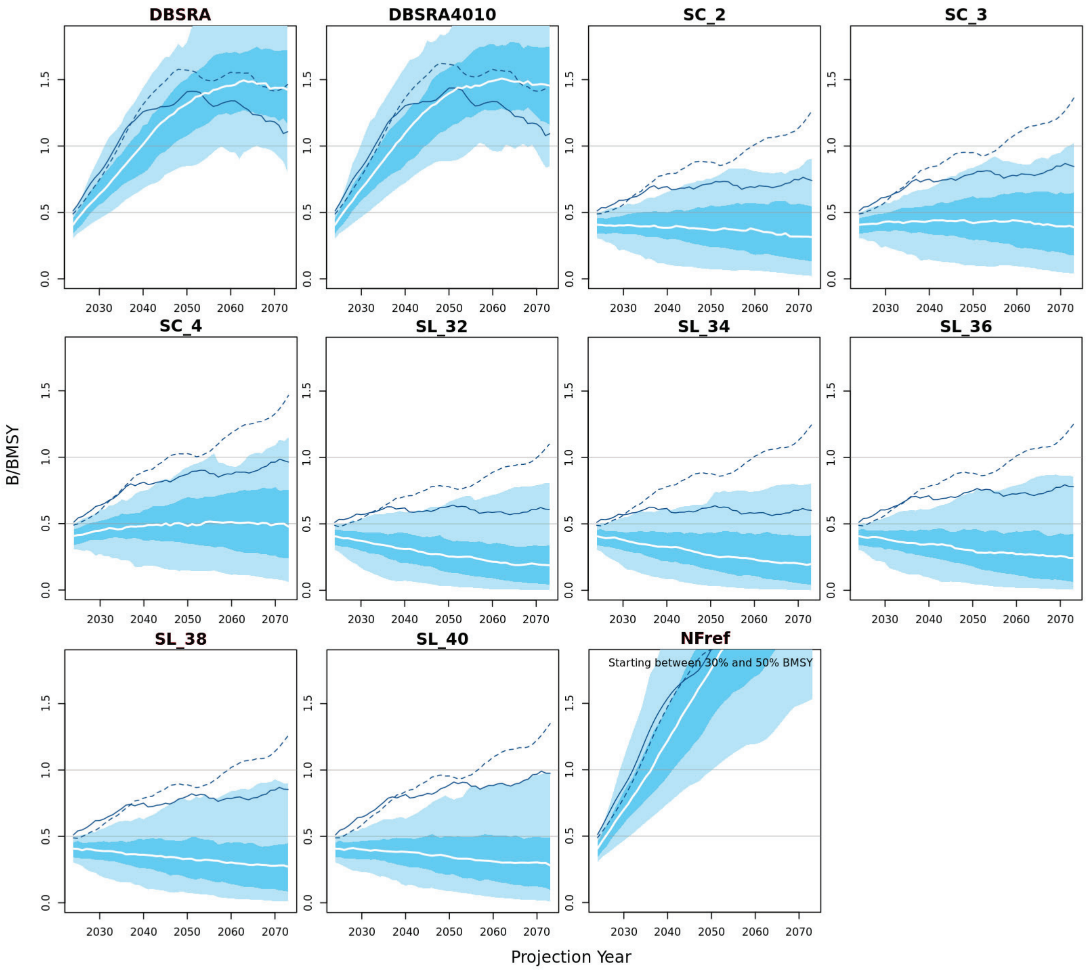

Figure 6. Long-term (50 years) biomass projection under a stock rebuilding scenario (started from the stock condition at 30 to 50% B_{MSY}) relative to the target and limit reference points (B/B_{MSY}) for the *P. leopardus*. The light blue represents a 90% probability interval, the dark blue represents a 50% probability interval, the white line is the median estimate, and the two dark blue lines are example simulations. The two horizontal grey lines represent target and limit reference points. DBSRA = depletion-based stock reduction analysis; DBSRA4010 = depletion-based stock reduction analysis with 40–10 rule; SC_2 to SC_4 = two to four months seasonal closure; SL_32 to SL-40 = size limit at 32 to 40 cm; NFref = no-fishing reference. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

3.4. Yield Projections

In general, the yield projection plots over a 50-year period (from 2024 to 2074) showed a very good performance of the 10 MPs for maintaining the yields higher, or stable, above the current yields, i.e., $Yt/Y_{curr} > 1$ (Figure 7). The seasonal closure and size limit MPs showed a stable yield projection above the current yield with relatively low uncertainty (shown by narrow probability intervals). In contrast, DBSRA and DBSRA4010 showed a very high yield projection with a wide range of probability intervals (high uncertainty). This suggests that selecting any of these MPs to manage *P. leopardus* will likely maintain the long-term fishery yield in Saleh Bay.

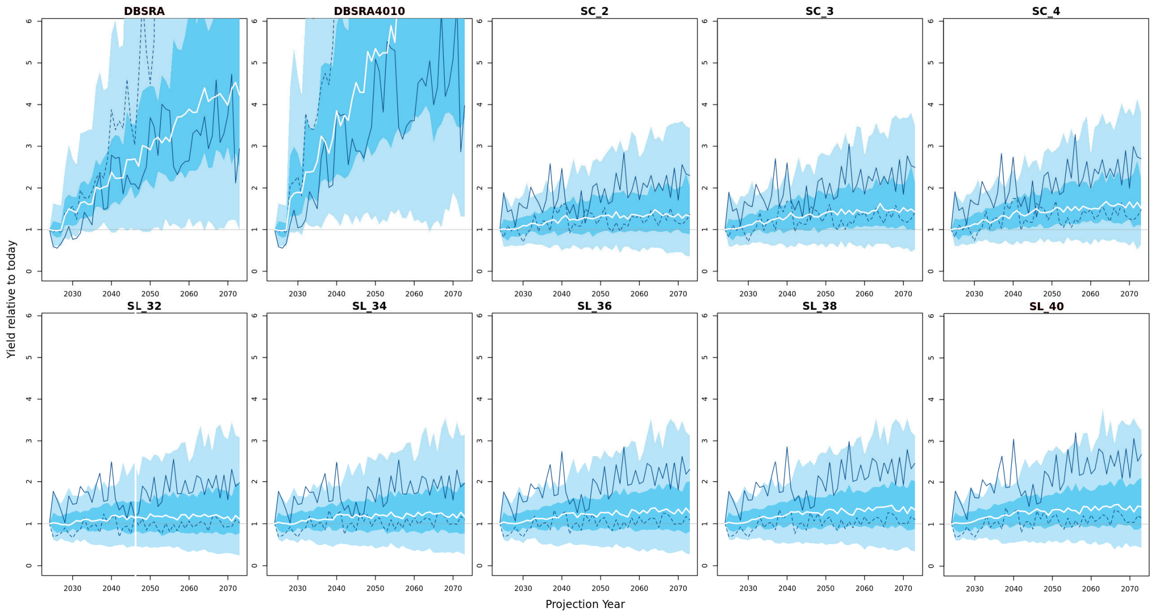

Figure 7. Yield projection relative to the current yield for the *P. leopardus* over a 50-year period. The light blue represents a 90% probability interval, the dark blue represents a 50% probability interval, the white line is the median estimate, and the two dark blue lines are example simulations. The horizontal grey lines represent the $Y_t/Y_{curr} = 1$. DBSRA = depletion-based stock reduction analysis; DBSRA4010 = depletion-based stock reduction analysis with 40–10 rule; SC_2 to SC_4 = two to four months seasonal closure; SL_32 to SL_40 = size limit at 32 to 40 cm. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

3.5. Sources of Uncertainty in Projections of the Long-Term-Yield

The sources of uncertainty contributing to the variation in the long-term yield projections (%LTY uncertainty) for *P. leopardus* varied across the 10 selected MPs. The maximum %LTY uncertainty ranged from 13 to 18% for the answers to the questionnaire (Figure 8), and were much higher (maximum = 27%) for the fishery data matrix (Figure 8). From the answers to the questionnaire, the longevity, fishing effort variation, TAC and TAE offsets, and hyperstability were the top five variables contributing to the uncertainty in projections of %LTY for the DBSRAs (Figure 7). The longevity (maximum age), steepness, fishing effort variation, post-release mortality, and hyperstability were the top five variables contributing to the uncertainty in projections of %LTY for the seasonal closure MPs, ranging from 11 to 14%. Fishing effort variation, post-release mortality, and hyperstability were also in the top five variables contributing most to the uncertainty in projections of %LTY for size limit MPs, as well as selectivity and future catchability. The uncertainty plots for the fishery data matrix are only available for MERA's default MPs (DBSRAs). The top five contributing variables to uncertainty in the projections for the %LTY from the fishery data matrix were biases in depletion, von Bertalanffy's *k*, length-at-maturity, and catch and depletion error (Figure 9). Depletion bias contributed by far the greatest to long-term yield uncertainty, i.e., 22%, in these projections, compared with <15% for all other variables (Figure 9).

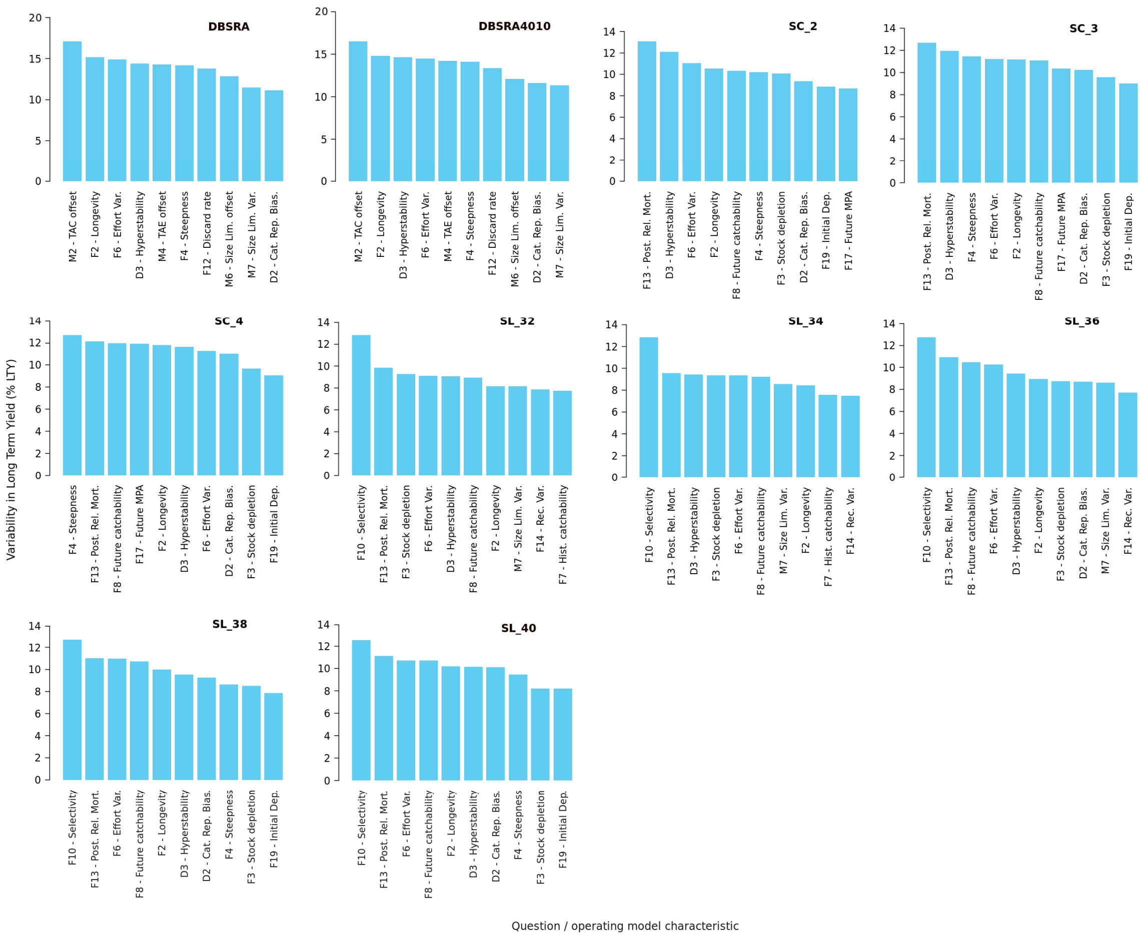

Figure 8. Source of uncertainties of 10 selected management procedures (MPs) for projections of the long-term yield for *P. leopardus* in Saleh Bay, West Nusa Tenggara, Indonesia, sourced from the answers to the MERA questionnaire. Some abbreviated variables: Effort Var. = effort variation; Size Lim Var. = size limit variation; Cat. Rep. Bias. = catch reporting bias; Post. Rel. Mort. = post-release mortality; Initial Dep. = initial depletion; Rec. Var. = recruitment variability; Hist. catchability = historical fishing efficiency. Descriptions of MPs are given in Table 2 and Supplementary Material S3.

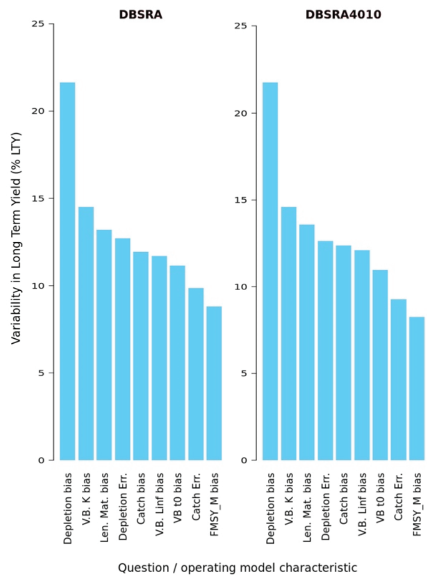

Figure 9. Source of uncertainties of the two default management procedures for projections of the long-term yield of *P. leopardus* in Saleh Bay, West Nusa Tenggara, Indonesia, from the fishery data matrix. Some abbreviated variables: VB. K. Bias = bias in von Bertalanffy's *k* variable estimation; Len. Mat. Bias = length maturity bias; Depletion Err. = depletion error; VB. Linf Bias = bias in von Bertalanffy's L_{inf} variable; catch Err. = historical catch error; FMSY_M bias = bias in F_{MSY}/M estimation. Descriptions of MPs are given in Table 2 and Supplementary Material S3. DBSRA = Depletion-based stock reduction analysis.

4. Discussion

Using the Method Evaluation and Risk Assessment (MERA) application, the effect of a suite of the 10 selected default and custom management procedures (MPs) on the short (10 years), medium (20 years), and long-term (46–50 years) biomass and yield of the *P. leopardus* stock in Salah Bay, Indonesia, were evaluated, and uncertainties in the simulations were determined. The management procedures were evaluated in two groups: (i) simulations based on current fishery conditions and (ii) simulations under the stock rebuilding scenario. Simulations to evaluate management procedures in this study were performed based on quantitative information to the MERA questionnaire and fishery data matrix, which contains data and information on fishery dynamics, management, biology and life history, and historical catch and effort.

4.1. Performance of Management Procedures

Of the four groups of MPs simulated in MERA, i.e., total allowable catch (TAC), total allowable effort, size limits, and spatial closures, the majority of TAC-based MPs showed a high probability of achieving the limit and target reference points for biomass. Custom seasonal closure MPs, ranging from two to four months, also showed relatively high probabilities of achieving these reference points. In contrast, the default size limit and spatial closure (MPA)-based MPs (input control) were simulated to have low probabilities of achieving the reference points for almost all scenarios. The high retention of fish smaller than the 50% size-at-maturity (L_{50}) likely contributed to the low performance of the size-limit-based MPs, especially in the stock rebuilding scenario. The catch-at-length data distribution in the fishery data matrix, collected from fish landing sites, suggests that almost 44% of *P. leopardus* catches are smaller than the estimated L_{50} (38.8 cm TL). Although the mean catch size of *P. leopardus* increased after enacting the size limit regulation [14], the current size limit regulation in Saleh Bay for *P. leopardus* is still nearly 7 cm smaller than the estimated L_{50}.

At least two input parameters contributed to the poor performance of the spatial closure or marine protected area (MPA)-based MPs: (i) the very small size of the existing no-take area (NTA) of two MPAs, causing (ii) high spatial mixing (movement) of this species in and out of the NTAs. The total area of the NTAs of existing MPAs in Saleh Bay (Pulau Liang-Ngali MPA and Pulau Lipan-Rakit MPA, area = 45.4 km^2) is ~2.2% of the total area of Saleh Bay. These two areas were designated as MPAs because they are the hotspots for coral reef ecosystem in Saleh Bay. The simulation results suggest that the current MPAs may not contribute to reducing fishing pressure on biomass. This may be because each of the NTAs of the two MPAs in Saleh Bay only has an area between 0.29 km^2 and 10.1 km^2 [36,37]; this is smaller than the home range of *P. leopardus* which can reach up to 28.2 km^2 [8,38,39]. Furthermore, the majority of grouper fishing activities in Saleh Bay are carried out outside the MPA [40]. Thus, the existing MPAs will likely have a very small impact on sustaining stock.

4.2. Selection of Management Procedures

The MP selection criteria resulted in 18 MPs that achieved limit and target reference points (LRP, TRP) for biomass and yield well. However, further analysis based on the trade-off plots showed that only two default MPs, both based on the depletion-based biomass stock reduction analyses (DBSRA and DBSRA4010), performed well in maintaining the biomass above the TRP, while producing a reasonably high yield, i.e., yield at MSY. Trade-offs associated with fisheries management have been a component of providing management advice since the development of quantitative approaches in fisheries science [41,42]. In the context of single-species fisheries, trade-offs are often quantified for various purposes, for example, trade-offs between (i) maintaining high long-term yield and the risk of the stock dropping below some biomass thresholds, (ii) variability of catch and average catch associated with the selection of harvest control rules, or (iii) rate of stock rebuilding and maintaining yield during the rebuilding period [42]. In the current study, the LRP for long-term yield (LTY) was set as >0.5 MSY as an additional MP selection criterion. In general, although most of the selected MPs for *P. leopardus* showed a good performance in maintaining biomass above the LRPs, only a few had a reasonably high probability of reaching LTY > 0.5 MSY.

The trade-off plot analysis suggests that only DBSRA and DBSRA4010 may be selected as management recommendations as they had the highest probabilities of achieving the biomass and yield targets. Nevertheless, the size limit and seasonal closure MPs may also be considered as management recommendations as they have a high probability of maintaining biomass above the limit reference point (LRP) and a high probability of maintaining a high yield. In certain fishery conditions (e.g., low stock biomass or spawning potential ratio), the fishery manager may consider implementing MPs with a high probability of keeping the stock biomass above its limit reference point and placing less weight on achieving the yield target, as an initial stage of fishery management.

The DBSRA methods assume long-term historical catch data from the beginning of the fishery are available [43,44], together with estimates of four key parameters: (1) the current depletion level, (2) the natural mortality rate (M), (3) the ratio of F_{MSY} to natural mortality (F_{MSY}/M), and (4) the biomass at MSY relative to the unfished biomass (B_{MSY}/B_0) [19]. The four key parameters used in this study were derived from a previous study on the *P. leopardus* stock in Saleh Bay [26] that relies on very short historical catch data (2016–2017), which may result in high uncertainty in the parameter estimation. The current study also used relatively short historical catch data, i.e., 2009–2021 for total catch and 2016–2021 for catch-at-length composition. In addition, the 2009–2021 historical total catch data were derived from the provincial fisheries statistics, and the 2016–2021 catch-at-length composition was derived from the fish landing monitoring program. The limited catch data and differences in sources for the historical catch data will also produce uncertainty in the simulation results. Although the available data are limited and may have high uncertainty, these data are the only available data for describing the historical condition of *P. leopardus* stocks and fishery in the study area.

Stock depletion (*D*) in this study is defined as the ratio between current and unfished biomass [19]. However, stock depletion is very difficult to estimate, especially in data-limited fisheries [45]. Historical catch data in Saleh Bay are available from 2009 to 2015 (fisheries statistics data) and 2016 to 2021 (FLM program data). Given no adequate long-term historical catch data (e.g., 30 years or more) are available for Saleh Bay, estimating the stock depletion using the available catch data (13 years) is problematic. Prior estimation of stock depletion using MERA's stock determination mode produces an estimated depletion value of 0.52 (spawning stock biomass; SSB = 0.52 B_0). This estimation result was not chosen based on two considerations: (1) the estimation is based on relatively short historical catch data, and (2) this value is too optimistic when referring to Amorim et al. [1], where SSB > 40% is a non-fully exploited stock, which is contradictory to the results of the previous studies in Saleh Bay [14,16].

Based on the above consideration, the current stock depletion value provided to MERA for model conditioning was estimated based on the current SPR (spawning potential ratio) value of *P. leopardus* in Saleh Bay, using the equilibrium MSY and SPR (spawning potential ratio) relationship [34] and the general equilibrium model for the biomass−MSY relationship (e.g., [35]). Based on the *D* value set by the user in MERA, for example, DBSRA generally estimates TAC based on the estimated value of F_{MSY} multiplied by the current estimate of abundance ($B_0 \times D$). The DBSRA models have been used by the Pacific Fishery Management Council to set and evaluate the Overfishing Limit (OFL) and Acceptable Biological Catch (ABC) for data-limited mackerel (Scombridae), butterfish (Stromateidae), snapper (Lutjanidae), porgy (Sparidae), sole (Pleuronectidae), and a wide range of rockfish (Sebastidae) species [19,43,46].

In practice, it is difficult to accurately estimate the exact value of the total allowable catch for *P. leopardus* based on the DBSRA methods. The TAC calculation for the DBSRA models relies on the assumed fishing mortality at MSY (F_{MSY}) and the estimation of the current stock biomass (virgin biomass multiplied by the estimated depletion rate). In addition, the current annual catch data are considered under-reported (it is estimated that only ~50% of the actual annual catch is recorded in the regular catch monitoring; where catch monitoring is only conducted at maximum 15 days per month), giving an even more challenging TAC estimation.

The custom MPs based on reducing catch proportionally, extended seasonal closures, and size limits were developed and tested to explore alternative MPs to those based on TAC. The closed-loop simulation results for the custom MPs suggest that reducing annual catch to 10 to 40% will maintain the biomass but with a relatively low probability of achieving LTY. On the other hand, reducing fishing effort through a seasonal fishing closure of 3 to 4 months suggests a better trade-off between LTY and stock biomass. In addition, setting a catch size limit at 40 cm for *P. leopardus* also showed a good yield–biomass trade-off. However, the feasibility of implementing a 3 to 4 month fishing closure and setting a minimum catch size of 40 cm, 8 cm above the current size limit, is not likely to gain the approval from stakeholders needed to be implemented effectively. Nevertheless, reducing the fishing effort through a seasonal closure consistently showed the best yield–biomass trade-off. These findings are consistent with a study by Williams and Shertzer [47], where controlling fishing mortality is likely more effective than gear selectivity in sustaining harvest and maximizing yield for species with low natural mortality, such as the *P. leopardus*. In addition, the current agreed size limit is about 7 cm below the estimated L_{50} of 38.8 cm (nearly 20% smaller than L_{50}); thus, exploring possible accepted larger size limit is worthwhile.

The "cost of uncertainty" calculation helps to understand which areas of current knowledge of the fisheries system need further investigation and the level of uncertainty that might occur when management advice is implemented [24]. For example, fishing effort variation, hyperstability, and estimation of von Bertalanffy's *k* and length-at-maturity are among the highest sources of uncertainty for the majority of the tested MPs; hence, these are the areas in the fishery system that need further study in Saleh Bay. The length-at-maturity is particularly important to help set accepted size limits that are likely to have considerable effects on the spawning stock and may be used to rebuild the spawning potential ratio [6]. Understanding the source of uncertainties also helps fisheries managers be aware of uncertainties (% of variability) in expected management outcomes (i.e., yield) when a particular management procedure is implemented.

4.3. Identification of Recommended Management Procedures

MSE simulations favor TAC-based management for the *P. leopardus* fishery in Saleh Bay over size limits and seasonal closures in reaching the target biomass reference point (TRP). However, the actual TAC number should be carefully defined based on the current annual catch, prior to implementation. This requires a better estimate of the current annual catch for setting an accurate TAC since the current estimate might not reflect the actual annual catch.

Learning from the application of MSE using data-limited methods (DLMs) for some key fisheries (e.g., barred sand bass, southern California halibut, southern red sea urchin, and warty sea cucumber in California [22]), this approach helped the authority and stakeholders evaluate and identify a range of acceptable management procedures specific for each fishery (e.g., effort control for barred sand bass, and output control for the southern California halibut) with a high probability of performing well over a range of stock and fishery system uncertainties. The MSE approach also helped identify the need for additional information to improve data collection and research programs, for example, the estimation of natural mortality rate of the warty sea cucumber [22]. The MERA application provides a generic DLM tool that is accessible online for wide use of MSE, with the potential for progression to tailored and more inclusion of data-rich methods. Furthermore, when needed, the MSE approach can also be used with customized DLM methods for specific fisheries. Carruthers [21] showed good examples of creating customized DLM methods specifically to meet the applicability to some Canadian data-limited fisheries (arrowtooth flounder, canary rockfish, and rougheye rockfish). There were 55 custom DLM tools developed for these fisheries that cover a wide range of functions, e.g., stock assessment, stochastic model for historical data reconstruction, and graphing tool to summarize simulation result [21].

Fisheries managers and stakeholders in Saleh Bay should understand the limitations of MERA, including the risk (cost of uncertainties) and behavior of each management procedure (MP). For example, the knowledge on species' maximum age (longevity) contributes the highest uncertainty to the estimation of long-term yield (LTY) for the depletion-based stock reduction analysis (DBSRA) MPs, while knowledge about catch selectivity contributes the highest uncertainty to the size limit MPs. Furthermore, the effectiveness of implementing size limit MPs varies among species with different reproductive strategies and lengths-at-maturity, but MERA implements the same treatment in the simulation regardless of the species' reproductive strategies [48]. *Plectropomus leopardus* is a protogynous hermaphrodite and a large, fecund species, and the size limit is probably more effective in protecting males and gives juveniles a greater chance of reaching reproductive age [49] compared to other species. In addition, a significant challenge in stock assessment (i.e., using MSY as a performance indicator) is the reliance on a single metric to describe fish population dynamics and their life histories. Furthermore, stock assessment methods are often built on simplified assumptions that fail to capture the inherent ecosystem variability, which creates uncertainties. When considering the distribution tails of population characteristics, the effects of these simplifications become clear, where the selection of distribution tails of population characteristics can significantly impact the robustness of the models used in the simulation. Thus, such limitations in MERA's operating model should also be acknowledged.

There have been many efforts to manage small-scale and data-limited fisheries, including through catch size limits, spawning seasonal closures, mesh size regulation, and marine protected area designation (e.g., [6,8]). Since 2018, the West Nusa Tenggara (WNT) government has also implemented grouper and snapper fisheries regulations for Saleh Bay in the form of size limits, fishing gear specifications, and marine protected area (MPA) management through Governor Regulation No. 32/2018 [15]. The effectiveness of limiting catch (TAC) and fishing effort (TAE) management approaches for snapper and grouper in Saleh Bay has not been specifically evaluated. The application of TAC and TAE is not currently considered feasible due to a lack of supporting regulations and management tools for implementing these measures and ensuring compliance with them.

Applying the TAC approach (e.g., by a quota system) requires developing additional supporting systems, such as a catch reporting mechanism for fishers and fish collectors. In addition, it is very important to have the best estimate of the real annual catch of the species, to ensure an accurate TAC setting. Thus, for the initial step, applying a TAC in Saleh Bay can only be initiated when: (i) the actual annual catch can be estimated accurately, and (ii) strong data collection and catch reporting mechanisms are in place. However, ensuring the fishers and fish collectors fully comply with the TAC (i.e., to halt fishing activities when the TAC is reached) is a very challenging task given the dynamic and complexity of the fisheries. Potential mechanisms for making TAC measures feasible in this region include: (i) designing and implementing a surveillance and enforcement mechanism to ensure TAC compliance and/or (ii) implementing an inclusive management policy by ensuring the full participation of fishers and fisheries businesses (particularly processors) in monitoring, evaluation, and management decision making.

Reducing fishing effort through seasonal closures is also challenging and is unlikely to be accepted by stakeholders, especially if the closure is implemented over a relatively long period of time (e.g., 2–4 months). Another challenge in reducing fishing effort in small-scale fisheries is the limited alternative livelihood options for fishers [8]. However, more targeted seasonal closures can be implemented in relation to the spawning season for specific species. For example, implementing spatial and temporal fishing restrictions at spawning aggregation sites during the spawning season will significantly reduce pressure on grouper populations [50,51]. In addition, banning sales of vulnerable grouper species during the closed season may also be an option to amplify the effectiveness of fishing pressure reduction [3]. However, implementing seasonal closures will likely cause fishers to change their target species, which may shift pressure to other fish populations. This phenomenon was reported by Chavarro et al. [52], where a four-month closure of grouper spawning aggregation sites during the spawning season in Palau led to potentially overfishing on bluespine unicornfish (*Naso unicornis*) by the spear fishery. Therefore, the feasibility of implementing seasonal closures needs to be carefully studied and evaluated.

Since 2021, the government of Indonesia has been implementing a new fishery policy called "perikanan terukur" (measurable fisheries), which includes implementing a catch quota system. This policy provides both an opportunity and a challenge for how the government can provide regulations and develop a system that allows the implementation of a quota system for industrial-scale and small-scale fisheries. Ensuring compliance with the quota system in small-scale fishery is undoubtedly more challenging than for large-scale/industrial fisheries because of (1) the dynamic characteristics of the SSF, (2) SSF do not have a vessel licensing system, (3) the majority of catches from SSF are not landed at official fishing ports, and (4) the logbook system to record catch and effort has not been applied to SSF in Indonesia (see also [53–55]).

An earlier study conducted on the stock status of *P. leopardus* in Saleh Bay [5] revealed that the spawning potential ratio (SPR) for this species is notably low, falling below the reference point threshold (SPR < 0.2). However, a recent assessment of fisheries management in Saleh Bay has indicated an improvement in the stock status, with SPR values exceeding 0.2 [16,33]. This improvement can plausibly be attributed to the enforcement of fishing regulations enacted in 2018. Nevertheless, this study implies that the size limit regulation and the existing no-take areas within the marine protected area (MPA) in Saleh Bay may not be effectively accomplishing the targeted biomass reference points. The relatively low compliance with the size-limit regulation (also consistent with Effendi et al. [14]), and the very small proportion of MPA's no-take area compared to the total fishing ground areas in Saleh Bay [40], is likely to contribute to the poor performances of these MPs. There are several approaches that can be taken to improve fishers' compliance with regulations, including (1) strengthening the control to fishing effort and unsustainable fishing practices (e.g., [8]), (2) enforcing rules through stricter sanctions (e.g., [53]), (3) improving fishers' understanding by involving them in catch monitoring activities (e.g., [6,56]), and (4) strengthening the participation of fishing communities in fisheries planning and management (e.g., [57]).

Numerous studies have shown that marine protected areas are effective in protecting highly fecund fish, such as *P. leopardus*, in maintaining their reproductive outputs [58], reducing fishing pressure [59], and protecting important habitats for different stages in the life cycle such as nursery, feeding, and spawning grounds [60]. A study by Williamson et al. [61] found that *P. leopardus* juveniles could spread as far as 250 km from their natal area. This indicates that the role of MPA is very important to protect the reproductive cycle of this species. If managed effectively, MPAs will provide spill-over effects, increasing fish biomass in surrounding waters [61,62]. Conversely, MPA management solely is unlikely to be effective in sustaining fisheries stock [62]; hence, a combination with market-based management approaches and property rights systems is recommended as a complement to conventional fisheries management through limiting catch and effort [8,63].

The MPA's no-take areas in Saleh Bay were established based on considerations of coral reef health and reef fish abundance and biomass [36,37], and were not specifically designed to maintain specific fisheries stocks. Furthermore, limiting fishing efforts through increasing the size of spatial closures (MPA) in Saleh Bay is unlikely to improve in the near future, since the existing marine spatial plan of West Nusa Tenggara (Provincial Regulation No. 12 of 2017) does not allocate areas for establishing new MPAs in Saleh Bay, in addition to the existing two MPAs. Nevertheless, the expansion of no-take areas within the existing MPAs may also be recommended and implemented in the future during the regular five-year MPA zoning plan evaluation. In addition, the effectiveness of surveillance and law enforcement to ensure compliance with the existing MPA no-take areas in Saleh Bay remains uncertain (see [40]); this is the area where the management authority should prioritize their management efforts. Thus, appropriate zoning design combined with enduring enforcement and compliance will generate effective MPAs to achieve conservation and fisheries management goals [64].

Although the size limit and spatial closure MPs have poor performances in achieving the desired biomass target, these MPs show good performances in maintaining biomass above the limit reference point. In addition, these MPs also show a very good performance in maintaining a stable long-term yield equal to today's yield. Given that the TAC management procedures are still difficult to implement, implementing size limits and spatial closures through MPA management remains the best option in the current management capacity and fisheries condition in Saleh Bay. However, it is still important for the West Nusa Tenggara government to continue to improve the implementation of the existing fisheries management plan in Saleh Bay while simultaneously developing the necessary supporting management instruments (i.e., policy, regulations, and mechanisms) toward implementing TAC and effort control measures if the rebuilding stock policy for *P. leopardus* is implemented. Overall, the results from this study are not the final advice to management, as they need to be discussed with a wide range of stakeholders before being translated into management policy.

The application of the MERA tool to evaluate management procedures in this study is built on knowledge of stock conditions based on historical catches and biological characteristics of the species studied. However, from an ecosystem dynamics point of view, stock abundance will also be influenced by habitat changes (e.g., nursery capacity) caused by dynamic interactions between biotic and environmental components (e.g., [65]), which is outside the scope of this study. Further study to understand the interactions between stocks and the habitat and environmental changes will be critical to reducing uncertainty in future stock estimates and improving adaptive fisheries management in Saleh Bay.

5. Conclusions

This study has demonstrated how the Method Evaluation and Risk Assessment platform can be used to evaluate currently implemented management procedures and how likely they are to achieve desired management objectives. Although the simulation results showed that, if implemented and rigorously enforced, the current management procedures (minimum catch size and spatial closure) in Saleh Bay are capable of keeping the stock biomass of *P. leopardus* above the limit reference point, the management authority needs to explore at least three options in order to achieve the management objective of increasing the biomass of this species to $B/B_{MSY} > 1$. These are (1) reducing the current fishing pressure through the implementation of TAC and seasonal closure, (2) improving the management effectiveness (monitoring and enforcement) of the currently implemented management procedures, and (3) improving management plans of existing MPAs to support fisheries management objectives.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/fishes8100498/s1, Table S1: Data input to MERA for *Plectropomus leopardus*; Table S2: Fishery data matrix for operational model conditioning; Table S3: The 20 MERA default management procedures (MPs) simulated in this study; Script S4: R-scripts for custom management procedures.

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References

1. Amorim, P.; Sousa, P.; Jardim, E.; Menezes, G.M. Sustainability Status of Data-Limited Fisheries: Global Challenges for Snapper and Grouper. *Front. Mar. Sci.* **2019**, *6*, 654. [CrossRef]
2. Amorim, P.; Westmeyer, M. *Snapper and Grouper: SFP Fisheries Sustainability Overview 2015*; Sustainable Fisheries Partnership: Honolulu, HI, USA, 2015. Available online: http://sfpcms.sustainablefish.org.s3.amazonaws.com/2016/03/07/SFP_Sectoral_Report_Snappers_Groupers_Final-99f93c03-d0ade6ae.pdf (accessed on 12 January 2023).
3. Sadovy de Mitcheson, Y.J.; Linardich, C.; Barreiros, J.P.; Ralph, G.M.; Aguilar-Perera, A.; Afonso, P.; Erisman, B.E.; Pollard, D.A.; Fennessy, S.T.; Bertoncini, A.A.; et al. Valuable but Vulnerable: Over-Fishing and Under-Management Continue to Threaten Groupers so What Now? *Mar. Policy* **2020**, *116*, 103909. [CrossRef]
4. FAO. Food and Agriculture Organization of the United Nations Fishery and Aquaculture Statistics; Global Capture Production 1950–2013; Updated 2022. Available online: https://www.fao.org/fishery/statistics-query/en/global_production/global_production_quantity (accessed on 22 March 2023).

5. Halim, A.; Loneragan, N.R.; Wiryawan, B.; Hordyk, A.R.; Sondita, M.F.A.; Yulianto, I. Evaluating Data-Limited Fisheries for Grouper (Serranidae) and Snapper (Lutjanidae) in the Coral Triangle, Eastern Indonesia. *Reg. Stud. Mar. Sci.* **2020**, *38*, 101388. [CrossRef]
6. Prince, J.; Victor, S.; Kloulchad, V.; Hordyk, A. Length Based SPR Assessment of Eleven Indo-Pacific Coral Reef Fish Populations in Palau. *Fish. Res.* **2015**, *171*, 42–58. [CrossRef]
7. Mudjirahayu; Bawole, R.; Rembet, U.N.W.J.; Ananta, A.S.; Runtuboi, F.; Sala, R. Growth, Mortality and Exploitation Rate of *Plectropomus maculatus* and *P. oligocanthus* (Groupers, Serranidae) on Cenderawasih Bay National Park, Indonesia. *Egypt. J. Aquat. Res.* **2017**, *43*, 213–218. [CrossRef]
8. Frisch, A.J.; Cameron, D.S.; Pratchett, M.S.; Williamson, D.H.; Williams, A.J.; Reynolds, A.D.; Hoey, A.S.; Rizzari, J.R.; Evans, L.; Kerrigan, B.; et al. Key Aspects of the Biology, Fisheries and Management of Coral Grouper. *Rev. Fish Biol. Fish.* **2016**, *26*, 303–325. [CrossRef]
9. Andamari, R.; Moria, S.B.; Permana, G.N. Aspects of Leopard Coral Grouper (*Plectropomus leopardus*). *Indones. Aquac. J.* **2007**, *2*, 51–57.
10. Craig, M.T.; de Mitcheson, Y.S.; Heemstra, P.C. *Groupers of the World: A Field and Market Guide*; CRC Press: Boca Raton, FL, USA, 2011; ISBN 978-1-920033-11-8.
11. Herdiana, Y.; Agustina, S.; Hartati, I.D.; Utami, A.; de la Rosa, E.; Yulianto, I. Fisheries Management for the Leopard Coral Grouper (*Plectropomus leopardus*) in Saleh Bay, West Nusa Tenggara, Indonesia. In *Proceedings from Workshops on Management Strategy Evaluation of Data-Limited Fisheries: Towards Sustainability—Applying the Method Evaluation and Risk Assessment Tool to Seven Indonesian Fisheries*; Murdoch University: Perth, WA, Australia; IPB University: Bogor, Indonesia, 2021; pp. 129–150, ISBN 978-0-646-82951-7.
12. Ministry of Marine Affairs and Fisheries. Pusat Data Statistik dan Informasi (Pusdatin) Data Produksi Perikanan 2023. Available online: https://statistik.kkp.go.id/home.php (accessed on 12 August 2022).
13. Yulianto, I.; Kartawijaya, T.; Rafandi, M.T.; Agustina, S.; Pingkan, J.; Aminollah; Nurjamil; Sabariyono; Widodo; Ramadhan, S.; et al. *Profil Perikanan Tangkap di Teluk Saleh dan Teluk Cempi Provinsi Nusa Tenggara Barat*; Wildlife Conservation Society: Bogor, Indonesia, 2016.
14. Efendi, D.S.; Adrianto, L.; Yonvitner; Wardiatno, Y. An Evaluation of Grouper and Snapper Fisheries Management Policy in Saleh Bay, Indonesia. *IOP Conf. Ser. Earth Environ. Sci.* **2021**, *744*, 012013. [CrossRef]
15. Retnoningtyas, H.; Yulianto, I.; Soemodinoto, A.; Herdiana, Y.; Kartawijaya, T.; Natsir, M.; Haryanto, J.T. Stakeholder Participation in Management Planning for Grouper and Snapper Fisheries in West Nusa Tenggara Province, Indonesia. *Mar. Policy* **2021**, *128*, 104452. [CrossRef]
16. Agustina, S.; Natsir, M.; Gigentika, S.; Marliana, I.; Anas, A.; Rafandi, M.T.; Hernawati; Kartawijaya, T.; Prasetia, R. *Telaah Ilmiah Pengelolaan Perikanan Kerapu Dan Kakap Berkelanjutan Di Nusa Tenggara Barat 2023*; Forum Ilmiah Pengelolaan Perikanan Berkelanjutan (FIP2B Provinsi Nusa Tenggara Barat): Mataram, Indonesia, 2023.
17. Fisheries Improvement Program. Saleh Bay Snapper and Grouper—Bottom Longline and Handline. Available online: https://fisheryprogress.org/fip-profile/indonesia-saleh-bay-snapper-and-grouper-bottom-longline-and-handline (accessed on 22 March 2023).
18. Punt, A.E.; Butterworth, D.S.; de Moor, C.L.; De Oliveira, J.A.A.; Haddon, M. Management Strategy Evaluation: Best Practices. *Fish Fish.* **2016**, *17*, 303–334. [CrossRef]
19. Carruthers, T.R.; Punt, A.E.; Walters, C.J.; MacCall, A.; McAllister, M.K.; Dick, E.J.; Cope, J. Evaluating Methods for Setting Catch Limits in Data-Limited Fisheries. *Fish. Res.* **2014**, *153*, 48–68. [CrossRef]
20. Goethel, D.R.; Lucey, S.M.; Berger, A.M.; Gaichas, S.K.; Karp, M.A.; Lynch, P.D.; Walter, J.F. Recent Advances in Management Strategy Evaluation: Introduction to the Special Issue "Under Pressure: Addressing Fisheries Challenges with Management Strategy Evaluation". *Can. J. Fish. Aquat. Sci.* **2019**, *76*, 1689–1696. [CrossRef]
21. Carruthers, T.R. *Customizing DLMtool for Canadian Fisheries*; Technical Report for Fisheries and Oceans Canada; Fisheries and Oceans Canada: Vancouver, BC, Canada, 2017. Available online: https://www.datalimitedtoolkit.org/wp-content/uploads/2017/09/Customizing-DLMtool-for-Canadian-Fisheries-WEB.pdf (accessed on 25 March 2023).
22. Hordyk, A.; Newman, D.; Carruthers, T.R.; Suatoni, L. Applying Management Strategy Evaluation to California Fisheries: Case Studies and Recommendations. 2017. Available online: https://www.datalimitedtoolkit.org/wp-content/uploads/2017/07/Applying-MSE-to-CA-Fisheries-Case-Studies-Recommendations.pdf (accessed on 25 March 2023).
23. Loneragan, N.; Wiryawan, B.; Hordyk, A.; Halim, A.; Proctor, C.; Satria, F.; Yulianto, I. *Proceedings from Workshops on Management Strategy Evaluation of Data-Limited Fisheries: Towards Sustainability—Applying the Method Evaluation and Risk Assessment Tool to Seven Indonesian Fisheries*; Murdoch University: Perth, WA, Australia; IPB University: Bogor, Indonesia, 2021; ISBN 978-0-646-82951-7.
24. Carruthers, T.R.; Huynh, Q.C.; Hordyk, A.R.; Newman, D.; Smith, A.D.M.; Sainsbury, K.J.; Stokes, K.; Morison, A.; Agnew, D.; Parma, A.; et al. Method Evaluation and Risk Assessment: A Framework for Evaluating Management Strategies for Data-limited Fisheries. *Fish Fish.* **2023**, *24*, 279–296. [CrossRef]
25. Hordyk, A.R.; Carruthers, T.R. A Quantitative Evaluation of a Qualitative Risk Assessment Framework: Examining the Assumptions and Predictions of the Productivity Susceptibility Analysis (PSA). *PloS ONE* **2018**, *13*, e0198298. [CrossRef] [PubMed]

26. Agustina, S.; Panggabean, A.S.; Natsir, M.; Retroningtyas, H.; Yulianto, I. Yield-per-Recruit Modeling as Biological Reference Points to Provide Fisheries Management of Leopard Coral Grouper (*Plectropomus leopardus*) in Saleh Bay, West Nusa Tenggara. *IOP Conf. Ser. Earth Environ. Sci.* **2019**, *278*, 012005. [CrossRef]
27. Yulius, Y.; Ramdhan, M.; Ardiansyah, A. Characteristic of Saleh Bay Coastline. *Segara* **2018**, *14*, 117. [CrossRef]
28. Khasanah, M.; Nurdin, N.; Sadovy de Mitcheson, Y.; Jompa, J. Management of the Grouper Export Trade in Indonesia. *Rev. Fish. Sci. Aquac.* **2020**, *28*, 1–15. [CrossRef]
29. Khasanah, M.; Kadir, N.N.; Jompa, J. Reproductive Biology of Three Important Threatened/Near-Threatened Groupers (*Plectropomus leopardus*, *Epinephelus polyphekadion* and *Plectropomus areolatus*) in Eastern Indonesia and Implications for Management. *Animals* **2019**, *9*, 643. [CrossRef]
30. Ebisawa, A. Life History Traits of Leopard Coralgrouper *Plectropomus leopardus* in the Okinawa Islands, Southwestern Japan. *Fish. Sci.* **2013**, *79*, 911–921. [CrossRef]
31. Mathews, C.P.; Samuel, M. Growth, Mortality and Assessment for Groupers from Kuwait. *Kuwait Bull. Mar. Sci.* **1997**, *9*, 173–191.
32. Efendi, D.S.; Daud, A.R.; Utomo, D.S.; Muawanah, U.; Adrianto, L. Dampak kebijakan perikanan kerapu dan kakap terhadap keberlanjutan stok sumber daya di Teluk Saleh. In *Prosiding Seminar Nasional Riset dan Kebijakan Sosial Ekonomi Kalautan dan Perikanan Tahun 2020*; AMAFRAD Press: Jakarta, Indonesia, 2020; pp. 153–162. ISBN 978-623-7651-68-0.
33. Agustina, S.; Natsir, M.; Panggabean, A.S.; Hartati, I.D.; Rosdiana, A.; Qonita, Y.; Rafandi, M.T.; Hernawati; Yulianto, I. *Laporan Teknis: Evaluasi Implementasi Rencana Aksi Pengelolaan Perikanan Kerapu dan Kakap Berkelanjutan (P2K2B) Provinsi Nusa Tenggara Barat Tahun ke-2 2021*; Wildlife Conservation Society: Bogor, Indonesia, 2022.
34. Goethel, D.R.; Smith, M.W.; Cass-Calay, S.L.; Porch, C.E. Establishing Stock Status Determination Criteria for Fisheries with High Discards and Uncertain Recruitment. *N. Am. J. Fish. Manag.* **2018**, *38*, 120–139. [CrossRef]
35. Hoshino, E.; Pascoe, S.; Hutton, T.; Kompas, T.; Yamazaki, S. Estimating Maximum Economic Yield in Multispecies Fisheries: A Review. *Rev. Fish. Biol. Fish.* **2018**, *28*, 261–276. [CrossRef]
36. DKP. *Dinas Kelautan dan Perikanan. Rencana Pengelolaan dan Zonasi Kawasan Konservasi Perairan, Suaka Alam Perairan Pulau Lipan dan Pulau Rakit Kabupaten Sumbawa, Provinsi Nusa Tenggara Barat 2018*; Dinas Kelautan dan Perikanan Provinsi Nusa Tenggara Barat: Mataram, Indonesia, 2018.
37. DKP. *Dinas Kelautan dan Perikanan. Rencana Pengelolaaan dan Zonasi Taman Wisata Perairan Pulau Liang dan Pulau Ngali 2018–2038*; Dinas Kelautan dan Perikanan Provinsi Nusa Tenggara Barat: Mataram, Indonesia, 2018.
38. Matley, J.; Heupel, M.; Simpfendorfer, C. Depth and Space Use of Leopard Coralgrouper *Plectropomus leopardus* Using Passive Acoustic Tracking. *Mar. Ecol. Prog. Ser.* **2015**, *521*, 201–216. [CrossRef]
39. Zeller, D. Home Range and Activity Patterns of the Coral Trout *Plectropomus leopardus* (Serranidae). *Mar. Ecol. Prog. Ser.* **1997**, *154*, 65–77. [CrossRef]
40. Natsir, M.; Ruchimat, T.; Agustina, S.; Yulianto, I. Application of Global Positioning System Tracker to Detect the Fishing Ground Location and Effective Effort in Artisanal Fishery. *Sens. Mater.* **2019**, *31*, 803. [CrossRef]
41. Dowling, N.A.; Dichmont, C.M.; Venables, W.; Smith, A.D.M.; Smith, D.C.; Power, D.; Galeano, D. From Low- to High-Value Fisheries: Is It Possible to Quantify the Trade-off between Management Cost, Risk and Catch? *Mar. Policy* **2013**, *40*, 41–52. [CrossRef]
42. Punt, A.E. Strategic Management Decision-Making in a Complex World: Quantifying, Understanding, and Using Trade-Offs. *ICES J. Mar. Sci.* **2017**, *74*, 499–510. [CrossRef]
43. Dick, E.J.; MacCall, A.D. Depletion-Based Stock Reduction Analysis: A Catch-Based Method for Determining Sustainable Yields for Data-Poor Fish Stocks. *Fish. Res.* **2011**, *110*, 331–341. [CrossRef]
44. Carruthers, T.R.; Huynh, Q.; Hordyk, A. Data-Limited Methods Reference (DLMtool Functions 6.0.6). 2022. Available online: https://dlmtool.openmse.com/ (accessed on 15 July 2023).
45. Zhou, S.; Punt, A.E.; Ye, Y.; Ellis, N.; Dichmont, C.M.; Haddon, M.; Smith, D.C.; Smith, A.D. Estimating Stock Depletion Level from Patterns of Catch History. *Fish Fish.* **2017**, *18*, 742–751. [CrossRef]
46. Pacific Fishery Management Council. *Pacific Coast Groundfish Fishery Management Plan for the California, Oregon, and Washington Groundfish Fishery*; Pacific Fishery Management Council: Portland, OR, USA, 2022. Available online: https://www.pcouncil.org/documents/2022/08/pacific-coast-groundfish-fishery-management-plan.pdf/ (accessed on 15 July 2023).
47. Williams, E.H.; Shertzer, K.W. Implications of Life-History Invariants for Biological Reference Points Used in Fishery Management. *Can. J. Fish. Aquat. Sci.* **2003**, *60*, 11. [CrossRef]
48. Carruthers, T.R.; Hordyk, A.R. The Data-Limited Methods Toolkit (DLM Tool): An R Package for Informing Management of Data-limited Populations. *Methods Ecol. Evol.* **2018**, *9*, 2388–2395. [CrossRef]
49. Sadovy De Mitcheson, Y.; Craig, M.T.; Bertoncini, A.A.; Carpenter, K.E.; Cheung, W.W.L.; Choat, J.H.; Cornish, A.S.; Fennessy, S.T.; Ferreira, B.P.; Heemstra, P.C.; et al. Fishing Groupers Towards Extinction: A Global Assessment of Threats and Extinction Risks in a Billion Dollar Fishery. *Fish Fish.* **2013**, *14*, 119–136. [CrossRef]
50. Sadovy De Mitcheson, Y.; Colin, P.L.; Lindfield, S.J.; Bukurrou, A. A Decade of Monitoring an Indo-Pacific Grouper Spawning Aggregation: Benefits of Protection and Importance of Survey Design. *Front. Mar. Sci.* **2020**, *7*, 571878. [CrossRef]
51. Erisman, B.; Heyman, W.; Kobara, S.; Ezer, T.; Pittman, S.; Aburto-Oropeza, O.; Nemeth, R.S. Fish Spawning Aggregations: Where Well-placed Management Actions Can Yield Big Benefits for Fisheries and Conservation. *Fish Fish.* **2017**, *18*, 128–144. [CrossRef]

52. Bejarano Chavarro, S.; Mumby, P.J.; Golbuu, Y. Changes in the Spear Fishery of Herbivores Associated with Closed Grouper Season in Palau, Micronesia: Spear Fishery of Herbivores during Closed Grouper Season. *Anim. Conserv.* **2014**, *17*, 133–143. [CrossRef]
53. Lindley, J. Fishing Non-Compliance and Culture. *Mar. Policy* **2023**, *152*, 105581. [CrossRef]
54. Halim, A.; Wiryawan, B.; Loneragan, N.R.; Hordyk, A.; Sondita, M.F.A.; White, A.T.; Koeshendrajana, S.; Ruchimat, T.; Pomeroy, R.S.; Yuni, C. Developing a Functional Definition of Small-Scale Fisheries in Support of Marine Capture Fisheries Management in Indonesia. *Mar. Policy* **2019**, *100*, 238–248. [CrossRef]
55. Oyanedel, R.; Gelcich, S.; Milner-Gulland, E.J. Motivations for (Non-)Compliance with Conservation Rules by Small-scale Resource Users. *Conserv. Lett.* **2020**, *13*, 1120–1134. [CrossRef]
56. Prince, J.; Creech, S.; Madduppa, H.; Hordyk, A. Length Based Assessment of Spawning Potential Ratio in Data-Poor Fisheries for Blue Swimming Crab (*Portunus* spp.) in Sri Lanka and Indonesia: Implications for Sustainable Management. *Reg. Stud. Mar. Sci.* **2020**, *36*, 101309. [CrossRef]
57. Oyanedel, R.; Gelcich, S.; Milner-Gulland, E.J. A Synthesis of (Non-)Compliance Theories with Applications to Small-scale Fisheries Research and Practice. *Fish Fish.* **2020**, *21*, 1120–1134. [CrossRef]
58. Marshall, D.J.; Gaines, S.; Warner, R.; Barneche, D.R.; Bode, M. Underestimating the Benefits of Marine Protected Areas for the Replenishment of Fished Populations. *Front. Ecol. Env.* **2019**, *17*, 407–413. [CrossRef]
59. Hall, A.; Cameron, D.; Kingsford, M. Prohibiting Spearfishing Boosts Conservation Outcomes for Partially Protected Areas. *Biol. Conserv.* **2022**, *272*, 109662. [CrossRef]
60. Balbar, A.C.; Metaxas, A. The Current Application of Ecological Connectivity in the Design of Marine Protected Areas. *Glob. Ecol. Conserv.* **2019**, *17*, e00569. [CrossRef]
61. Williamson, D.H.; Harrison, H.B.; Almany, G.R.; Berumen, M.L.; Bode, M.; Bonin, M.C.; Choukroun, S.; Doherty, P.J.; Frisch, A.J.; Saenz-Agudelo, P.; et al. Large-scale, Multidirectional Larval Connectivity among Coral Reef Fish Populations in the Great Barrier Reef Marine Park. *Mol. Ecol.* **2016**, *25*, 6039–6054. [CrossRef] [PubMed]
62. Hilborn, R.; Micheli, F.; De Leo, G.A. Integrating Marine Protected Areas with Catch Regulation. *Can. J. Fish. Aquat. Sci.* **2006**, *63*, 642–649. [CrossRef]
63. Halim, A.; Loneragan, N.R.; Wiryawan, B.; Fujita, R.; Adhuri, D.S.; Hordyk, A.R.; Sondita, M.F.A. Transforming Traditional Management into Contemporary Territorial-Based Fisheries Management Rights for Small-Scale Fisheries in Indonesia. *Mar. Policy* **2020**, *116*, 103923. [CrossRef]
64. Edgar, G.J.; Stuart-Smith, R.D.; Willis, T.J.; Kininmonth, S.; Baker, S.C.; Banks, S.; Barrett, N.S.; Becerro, M.A.; Bernard, A.T.F.; Berkhout, J.; et al. Global Conservation Outcomes Depend on Marine Protected Areas with Five Key Features. *Nature* **2014**, *506*, 216–220. [CrossRef]
65. Kininmonth, S.; Blenckner, T.; Niiranen, S.; Watson, J.; Orio, A.; Casini, M.; Neuenfeldt, S.; Bartolino, V.; Hansson, M. Is Diversity the Missing Link in Coastal Fisheries Management? *Diversity* **2022**, *14*, 90. [CrossRef]

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Communication

First Record of the Red Cornetfish *Fistularia petimba* Lacepède, 1803 from Amorgos Island (Central Aegean Sea; Greece) and a Review of Its Current Distribution in the Mediterranean Sea

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Abstract: The rapid spread of non-native species (NNS) poses a significant threat to biodiversity globally, with the Mediterranean region being particularly susceptible due to increased human activities and its status as a marine biodiversity hotspot. In this study, we focus on the introduction and distribution of *Fistularia petimba*, a member of the Fistulariidae family, in the eastern Mediterranean Sea and a record from the coasts of Amorgos Island, Greece. Through a baseline fishery study conducted over 12 months, utilizing experimental sampling with gillnets, trammel nets, and longlines, one individual of *F. petimba* was captured off the coast of Katapola Bay. Morphological examination confirmed its identity, with measurements on meristic characteristics obtained and the stomach content analysed. This finding represents a significant addition to the documented distribution of *F. petimba* in the Mediterranean Sea, particularly in the Aegean Sea, underscoring the importance of ongoing research in uncovering new occurrences and expanding our understanding of marine biodiversity and ecosystem changes. Further investigation into the ecological preferences and population dynamics of *F. petimba* in the Aegean Sea is crucial for informed conservation and management efforts if this species is considered to be established.

Keywords: *Fistularia petimba*; red cornetfish; Mediterranean Sea; invasive species; non-native species; Lessepsian species; marine biodiversity; species distribution

Key Contribution: This study provides a review of a recent non-native species geographical distribution. This study provides insights on the crucial establishment phase for the species and signifies areas for further investigation to reveal life trait characteristics and succession rate.

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1. Introduction

One of the main threats that biodiversity currently faces is the rapid spread of non-native species (NNS). NNS are defined as an array of species spreading outside their natural or native distribution range [1]. Different areas worldwide have been experiencing vast impacts from the introduction of NNS, often related to increased human activities, such as the opening of canals, the continuous growth of the shipping industry across biogeographic barriers [2,3], a wide range of changes in water temperature due to climate change [4–6], fishing pressure [7–9], and habitat degradation or loss of species [10–12]. In the studied area of the Mediterranean Sea, recognized as one of the main hotspots of marine biodiversity [13,14], the effects of NNS are apparent, both in terms of introduction rate [15] and number of introduced species [16], leading to the global acknowledgment of the Mediterranean region as a hotspot area for NNS [17].

A region where visible changes in aquatic biodiversity have occurred is the eastern Mediterranean Sea, where a rapid introduction of fish species of Indo-Pacific origin are observed, i.e., the Levantine Sea [18–21], significantly raising the overall amount of fish biomass up to 90% in specific habitats [22,23]. These habitats include hard bottoms for *Siganus luridus* and *Siganus rivulatus*, and sandy bottoms and seagrass meadows for the *Lagocephalus sceleratus* [22]. Though Indo-Pacific fish species could potentially arrive through various ways in the Levantine Basin, they most likely arrive through immigration via the Suez Canal, which opened to shorten the commercial shipping ways between the Indian Ocean and the Mediterranean Sea in 1869 [24]. It is assumed that species that normally resided in the Red Sea and the Indian Ocean traversed through the Suez Canal and proceeded northwards along the Levant coast, actively or passively aided by human activity [25]. These species were named Lessepsian after the name of the constructor of the canal, engineer, and diplomat Ferdinand de Lesseps [26].

The aforementioned group of Lessepsian species established in the Mediterranean Sea currently includes *Fistularia commersonii* [27] and *Fistularia petimba*, also called cornetfishes or flutemouths, which belong to the Fistulariidae (order of Syngnathiformes). There is only one genus in this family, *Fistularia*, and four different species: *Fistularia commersonii* Rüppell, 1838; *Fistularia corneta* Gilbert and Starks, 1904; *Fistularia petimba* Lacepède, 1803; and *Fistularia tabacaria* Linnaeus, 1758 [28]. The species *F. tabacaria* inhabits the tropical Atlantic, while its closest relative *F. commersonii* inhabits the Pacific and Indian Oceans. *Fistularia petimba* spans the tropical Atlantic and Indo-West Pacific Oceans, whereas *F. corneta* is confined to the tropical eastern Pacific [29]. Fistularidae species are predators, inhabiting shallow waters of tropical and subtropical areas [29]. Although *F. commersonii* originated from the Indo-Pacific region [30], a wide geographical distribution has been observed in the eastern Mediterranean Sea [31], with multiple sightings of this Lessepsian immigrant. Due to its rapid growth and reproduction cycle, it has successfully formed large populations in the areas where it has been established, with notable ecological impacts on the native species [32]. *Fistularia commersonii* is a piscivorous species, mainly feeding on smaller fish and complementing its diet with some Crustacea species. As the size of the species increases, a corresponding increase in the size of prey consumed has been found [32].

The studied red cornetfish, *F. petimba*, is native to the Indo-West Pacific, the tropical Atlantic [30], and the East Atlantic Ocean [33]. With a time lag of 20 years since its first siting in the western Mediterranean Sea (1996), it has been reported in several locations in the eastern Mediterranean Sea over the last ten years [34,35]. In this study, we show the immigration path of *F. petimba* in the eastern Mediterranean Sea through a stepping stone process of establishment through the Suez Canal [36,37].

2. Materials and Methods

2.1. Sampling Methodology

A monthly experimental fishery sampling was performed in Amorgos Island, Central Aegean Sea, between September 2022 and August 2023 (Figure 1).

The sampling method used involved three types of gears: gillnets (GNSs), trammel nets (GTRs), and long lines (LLSs). The gears used were designed to study the population dynamics of targeted fisheries species.

For GNSs and GTRs, nine different mesh sizes (20, 22, 24, 26, 28, 30, 32, 36, 45 in mm) were used, and for LLS six different hook sizes (9, 10, 11, 12, 13, 14) were used to reflect the most commonly used mesh and hook sizes in small-scale fisheries of the Aegean Sea.

The nets used in this study had a total length of 900 m, consisting of 100-m panels for each of the nine mesh sizes. The height of each compartment was 100 meshes. The arrangement of mesh sizes for both GNSs and GTRs were randomly selected and are illustrated in Figures 2 and 3. For longlines, each compartment had a length of 200 m and was equipped with 200 hooks (100 hooks per hook size). One compartment comprised of hook sizes 9 and 10, and two more compartments comprised of combinations of hook sizes 11–12 and 13–14, respectively.

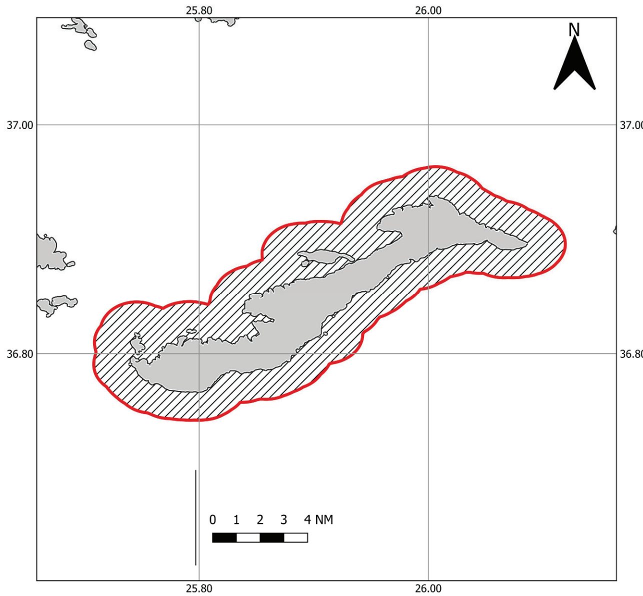

Figure 1. Map of the Amorgos Island, Central Aegean Sea, Greece. The red line marks the sampling area around Amorgos Island.

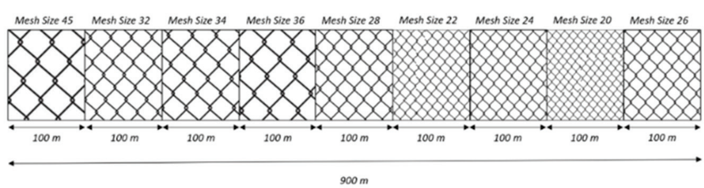

Figure 2. GNS mesh size arrangement in this study.

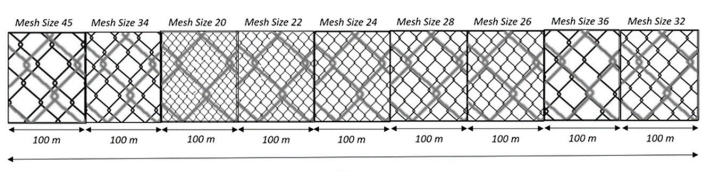

Figure 3. GTR mesh size arrangement in this study.

All species samples were stored in a freezer (−20 °C) until transportation and were deposited to the Laboratory of Applied Hydrobiology of the Agricultural University of Athens, Greece, for further examination.

2.2. Identification of the Species

Fistularia petimba was identified based on its morphological characteristics [29], following the genus description given by Fritzsche (1976): *Fistularia* species can be identified by their elongated body.

Fistularia petimba was distinguished from its confamilial species by its specific morphological features [29]: number of rays on the dorsal fin (13–17) and the anal fin (13–16), elongated bony plates embedded in the skin along the midline of the back, with posterior lateral line ossifications terminating in a retrorse spine (Figure 4).

(**a**)

(**b**)

Figure 4. *Fistularia petimba* individual from Amorgos Island and its identification characteristics: (**a**) reddish colour and (**b**) elongated bony plates embedded in the skin.

The morphometric characteristics shown in Table 1 were measured to the nearest second decimal in mm.

Table 1. Morphometric characteristics and measurements of *Fistularia petimba* from Amorgos Island, Greece.

Morphometrics	Measurement (mm)
Total Length without filament (TL)	395.00
Total Length with filament (TLf)	530.00
Filament Length (fL)	124.23
Standard Length (SL)	378.00
Fork Length (FL)	383.00
Body Deth (BD)	7.98
Head Length (HL)	142.00
Eye Diameter (ED)	10.97
Snout Length (SN)	114.00
Dorsal Fin Length (DFL)	15.41
Dorsal Fin Height (DFH)	28.89
Pectoral Fin Length (PFL)	6.66
Pelvic Fin Height (PFH)	16.81
Dorsal Fin Length (PvFL)	2.54
Pelvic Fin Height (PvFH)	6.83
Caudal Fin Length (CFL)	19.01
Caudal Fin Height (CFH)	5.10
Anal Fin Length (AFL)	15.16
Anal Fin Height (AFH)	27.37
Pre-dorsal Fin Length (pDFL)	67.00
Pre-pectoral Fin Length (pPFL)	319.00
Pre-pelvic Fin Length (pPvFL)	190.00
Pre-anal Fin Length (pAFL)	310.00

2.3. Distibution of F. petimba in the Mediterranean Sea

To compile a map with records of *F. petimba* in the Mediterranean Sea, a literature review (until February 2024) was performed using Google Scholar. This review used two main keywords, namely "*Fistularia petimba*" and "Red cornetfish", together with additional keywords to assure that *F. petimba* records were not missed: "Mediterranean Sea", "Invasive species", "NNS", "ecology", "habitat", and "lessepsian".

To visualize the species' geographical distribution in the Mediterranean Sea, a map illustrating the occurrences of *F. petimba* was generated using ArcGIS [38] and by integrating data from this study and published records from scientific journals.

3. Results

An individual of *F. petimba* (Figure 4) was captured using trammel nets (GTRs) with a mesh size of 26 mm, deployed between 24.5 m and 30.6 m depth on the 27th of May

2023 at 7:50 p.m. and hauled on the 28th of May 2023 at 6:40 a.m. (soak time 11 h and 50 min) off the coast of Katapola Bay, Amorgos Island, Greece (lon = “25.85879922” lat = “36.82760281”). The specimen had a total length of 395.00 mm and a total wet weight of 34 g. The measurements for each of the morphometric characteristics of the species are presented in Table 1.

4. Discussion

The increases in global trade and travel have also increased the chances for species to migrate, immigrate, or establish in areas beyond their native ranges, i.e., through widening and deepening canals or ballast water transport [39,40]. Immigration is most commonly associated with human intervention, such as the opening of canals. The Suez Canal, since its completion in the late 19th century, has served as a major conduit for the immigration of marine organisms between the Red Sea (and Indian Ocean) and the Mediterranean Sea. This artificial connection has facilitated the establishment of numerous NNS in the Mediterranean Basin, reshaping the region’s biodiversity and ecological dynamics [41,42]. When an area is invaded, it becomes a source for the subsequent spread of the organism to other locations in the basin [41]. Marinas in the Mediterranean Sea have been identified as significant areas for the establishment of NNS, not only for initial introductions, but also for subsequent secondary invasions, acting as stepping stones in the spread of NNS [40,43].

In the marine environment, NNS can become invasive, resulting in the displacement of native species, thus leading to a range of negative consequences. These effects encompass the loss of native genetic diversity, alterations to habitats, shifts in community composition, changes to food web dynamics and ecosystem functions, disruptions to the provision of ecosystem services, threats to human health, and significant economic damages [44].

The capture of *Fistularia petimba* in Amorgos Island (Central Aegean Sea) represents a significant addition to the existing knowledge of the species’ distribution in the Mediterranean region. Prior to this finding, 31 sightings were reported from the Mediterranean Sea, out of which only 1 was in the Aegean Sea, specifically in Samos Island (Figure 5, Table 2). The first record in Cadiz, Spain, appears to be an incidental catch from the Atlantic Ocean, while the subsequent occurrences, mainly in the eastern Mediterranean Sea, reveal a progressive invasion pattern from the Indo-Pacific region and the Red Sea into novel habitats. The review of its current distribution reveals its dispersal path along the coasts of Syria, Egypt, and Turkey. The species’ course is validated with further records from Cyprus and its possible establishment within the Aegean Sea. Due to the lack of available data on the potential impact of *F. petimba* on native species in the Mediterranean Sea, insights can be drawn from its closely related species, *Fistularia commersonii*. *Fistularia commersonii* is mainly piscivorous but also feeds on crustaceans [32]. It has a high reproductive rate and a prolonged spawning season extending from May to August, allowing for rapid population growth [45]. More than 70% of *Fistularia commersonii*’s diet includes economically valuable species. Its predation on small fish near the seabed, where they hatch and grow, disrupts ecosystem balance and reduces fish biomass [32]. This species quickly establishes itself in new environments, leading to significant ecological and economic consequences, including damage to fisheries (although gradually introduced to consumers) [32,45]. Given the similarities between *F. petimba* and *F. commersonii*, it is likely that *F. petimba* could have similar effects on the Mediterranean ecosystem. Effective monitoring and management strategies are crucial to mitigate these potential impacts. Each documented occurrence represents a critical point in the species’ biogeographic spread throughout the eastern Mediterranean Sea. This review of chronological records offers valuable insights into the stepping stone spread of *F. petimba* establishment in the Mediterranean Sea. Tracking its cross-border path from the Red Sea to the eastern Mediterranean Sea with subsequent records in the Aegean Sea provides valuable insights into the species’ establishment phase. As reproduction is a crucial part of the successful establishment of species in new areas, Papageorgiou et al. 2023 [46] estimated that the mean total length for gonad maturity was 440 mm for females and 410 mm for males. The results of Papageorgiou et al. 2023 [46] are in accordance with our results, wherein no visible gonads could be identified macroscopically.

Table 2. Data of the 32 validated records of *Fistularia petimba* in the Mediterranean Sea.

No	Location	Latitude	Longitude	Date (Capture)	Depth (m)	Gear Type	Sample Size
1	Cadiz, Spain [34]	36.455097	−4.703372	23 June 1996	50	Gillnet	1
2	Antalya Bay, Turkey [35]	36.793556	31.209167	28 October 2016	35–43	Bottom trawl	1
3	Ashdod, Israel [27]	31.813950	34.459717	12 November 2016	80	Bottom trawl	1
4	Antalya Bay, Turkey [35]	36.737417	31.434361	26 November 2016	30	Bottom trawl	1
5	Iskenderun, Turkey [35]	36.654400	36.186183	21 May 2017	35–38	Bottom trawl	2
6	Tripoli, Lebanon [47]	34.410000	35.770000	15 November 2017	N/A	Gillnet	1
7	Mersin Bay, Turkey [37]	36.128833	33.520667	22 November 2017	95	Bottom trawl	1
8	Antalya Bay, Turkey [37]	36.061867	32.534233	9 January 2018	70	Bottom trawl	2
9	Büyükeceli Coast (Mersin Bay) Turkey [36]	36.123139	33.467944	5 October 2018	150	Bottom trawl	2
10	Egypt [28]	El-Hamam—Sidi Kirayr.		9 March 2019	40–60	Bottom trawl	1
11	Lattakia, Syria [48]	35.518325	35.713492	29 July 2019	45	Gillnet	1
12	Lattakia, Syria [49]	35.243086	35.920000	24 September 2019	30	Gillnet	1
13	Gialia, Cyprus [50]	35.110000	32.490000	26 September 2019	55	Gillnet	1
14	Banyas, Syria [49]	35.518325	35.713492	29 September 2019	45	Gillnet	2
15	Gökova Bay, Turkey [51]	36.857889	27.896556	19 October 2019	15–20	Longline	1
16	Güllük Bay, Turkey [51]	36.857883	27.896561	17 November 2019	65	Bottom trawl	4
17	Cyprus [46]	34.747367	33.463400	14 July 2020	55	Bottom trawl	3
18	Cyprus [46]	34.964500	34.964500	15 July 2020	48	Bottom trawl	1
19	Cyprus [46]	34.759617	33.480650	16 July 2020	33	Bottom trawl	1
20	Cyprus [46]	34.924100	33.908050	24 July 2020	79	Bottom trawl	11
21	Cyprus [46]	35.081733	32.458700	24 July 2020	43	Bottom trawl	29
22	Cyprus [46]	34.635717	32.638517	27 March 2021	46	Bottom trawl	10
23	Cyprus [46]	34.661300	32.468650	27 March 2021	93	Bottom trawl	4
24	Bandirma Bay, Turkey [47]	40.416950	28.084000	11 June 2021	32	Trammel net	1
25	Cyprus [46]	34.693983	33.135567	4 August 2021	44	Bottom trawl	26
26	Samos, Greece [47]	37.706583	26.708783	7 November 2021	20	Trammel net	1
27	Cyprus [46]	35.060833	34.054383	8 August 2021	86	Bottom trawl	4
28	Cyprus [46]	34.750917	33.480933	8 August 2021	55	Bottom trawl	7
29	Cyprus [46]	34.750917	33.480933	8 August 2021	33	Bottom trawl	2
30	Cyprus [46]	34.692267	33.166750	8 August 2021	56	Bottom trawl	1
31	Cyprus [46]	34.699333	33.311817	13 September 2021	13	Trammel net	1
32	Amorgos, Greece (current study)	36.82760281	25.85879922	28 May 2023	24.5–30.6	Trammel net	1

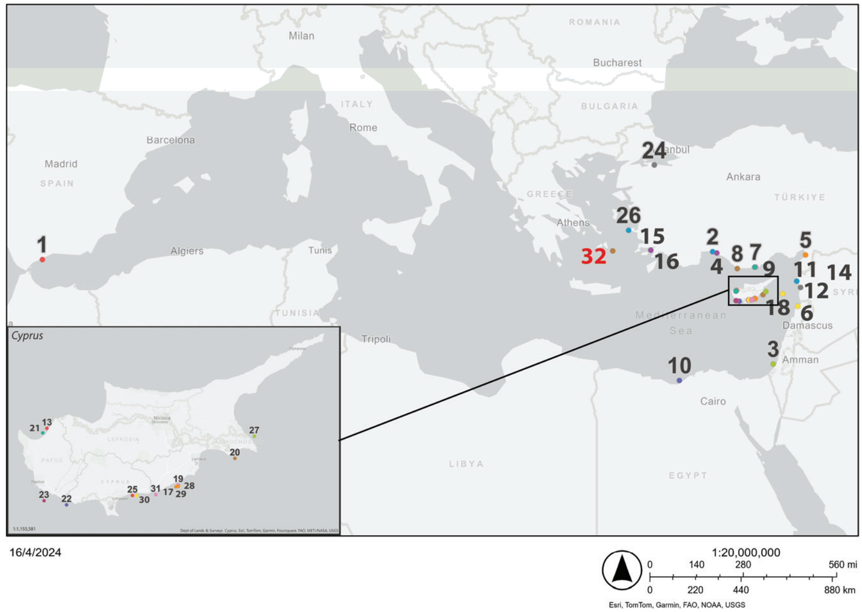

Figure 5. Records of *Fistularia petimba* in the Mediterranean Sea (Table 2) [38].

5. Conclusions

The occurrence of *F. petimba* in Amorgos Island suggests a wider presence within the Aegean Sea than previously recognized. This finding underscores the importance of ongoing fishery research and marine monitoring in uncovering new occurrences and expanding our understanding of marine biodiversity in the region. Further investigations into the ecological preferences, population dynamics, life traits, and potential impacts of *Fistularia petimba* in the Aegean Sea and the Mediterranean Basin are important to understand succession rates and ecological impacts, to enhance conservation efforts, and to inform sustainable management practices.

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References

1. European Environment Agency. Climate Change, Impacts and Vulnerability in Europe EEA. 2012. Available online: https://www.eea.europa.eu/publications/climate-impacts-and-vulnerability-2012 (accessed on 25 February 2024).
2. Lewis, S.L.; Maslin, M.A. Defining the Anthropocene. *Nature* **2015**, *519*, 171–180. [CrossRef] [PubMed]
3. Pyšek, P.; Hulme, P.E.; Simberloff, D.; Bacher, S.; Blackburn, T.M.; Carlton, J.T.; Dawson, W.; Essl, F.; Foxcroft, L.C.; Genovesi, P.; et al. Scientists' warning on invasive alien species. *Biol. Rev. Camb. Philos. Soc.* **2020**, *95*, 1511–1534. [CrossRef] [PubMed]
4. Genner, M.J.; Sims, D.W.; Wearmouth, V.J.; Southall, E.J.; Southward, A.J.; Henderson, P.A.; Hawkins, S.J. Regional climatic warming drives long-term community changes of British marine fish. *Proc. Biol. Sci.* **2004**, *271*, 655–661. [CrossRef] [PubMed]
5. Munday, P.L.; Jones, G.P.; Pratchett, M.S.; Williams, A.J. Climate change and the future for coral reef fishes. *Fish Fish.* **2008**, *9*, 261–285. [CrossRef]
6. Simpson, S.D.; Jennings, S.; Johnson, M.P.; Blanchard, J.L.; Schön, P.-J.; Sims, D.W.; Genner, M.J. Continental Shelf-Wide Response of a Fish Assemblage to Rapid Warming of the Sea. *Curr. Biol.* **2011**, *21*, 1565–1570. [CrossRef] [PubMed]
7. Pauly, D.; Christensen, V.; Dalsgaard, J.; Froese, R.; Torres, F. Fishing Down Marine Food Webs. *Science* **1998**, *279*, 860–863. [CrossRef] [PubMed]
8. Jennings, S.; Alvsvåg, J.; Cotter, A.J.R.; Ehrich, S.; Greenstreet, S.P.R.; Jarre-Teichmann, A.; Mergardt, N.; Rijnsdorp, A.D.; Smedstad, O. Fishing effects in northeast Atlantic shelf seas: Patterns in fishing effort, diversity and community structure. III. International trawling effort in the North Sea: An analysis of spatial and temporal trends. *Fish. Res.* **1999**, *40*, 125–134. [CrossRef]
9. Daan, N.; Gislason, H.; Pope, J.G.; Rice, J.C. Changes in the North Sea fish community: Evidence of indirect effects of fishing? *ICES J. Mar. Sci.* **2005**, *62*, 177–188. [CrossRef]
10. Jones, G.P.; McCormick, M.I.; Srinivasan, M.; Eagle, J.V. Coral decline threatens fish biodiversity in marine reserves. *Proc. Natl. Acad. Sci. USA* **2004**, *101*, 8251–8253. [CrossRef]
11. Airoldi, L.; Balata, D.; Beck, M.W. The Gray Zone: Relationships between habitat loss and marine diversity and their applications in conservation. *J. Exp. Mar. Biol. Ecol.* **2008**, *366*, 8–15. [CrossRef]
12. Wilson, S.K.; Fisher, R.; Pratchett, M.S.; Graham, N.A.J.; Dulvy, N.K.; Turner, R.A.; Cakacaka, A.; Polunin, N.V.C.; Rushton, S.P. Exploitation and habitat degradation as agents of change within coral reef fish communities. *Glob. Chang. Biol.* **2008**, *14*, 2796–2809. [CrossRef]
13. Myers, N.; Mittermeier, R.A.; Mittermeier, C.G.; da Fonseca, G.A.B.; Kent, J. Biodiversity hotspots for conservation priorities. *Nature* **2000**, *403*, 853–858. [CrossRef] [PubMed]
14. Coll, M.; Piroddi, C.; Steenbeek, J.; Kaschner, K.; Ben Rais Lasram, F.; Aguzzi, J.; Ballesteros, E.; Bianchi, C.N.; Corbera, J.; Dailianis, T.; et al. The biodiversity of the Mediterranean Sea: Estimates, patterns, and threats. *PLoS ONE* **2010**, *5*, e11842. [CrossRef]
15. Zenetos, A.; Gofas, S.; Verlaque, M.; Cinar, M.E.; Garcia Raso, J.E.; Bianchi, C.N.; Morri, C.; Azzurro, E.; Bilecenoglu, M.; Froglia, C.; et al. Alien species in the Mediterranean Sea by 2010. A contribution to the application of European Union's Marine Strategy Framework Directive (MSFD). Part I. Spatial distribution. *Mediterr. Mar. Sci.* **2010**, *11*, 381. [CrossRef]
16. Costello, M.J.; Coll, M.; Danovaro, R.; Halpin, P.; Ojaveer, H.; Miloslavich, P. A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. *PLoS ONE* **2010**, *5*, e12110. [CrossRef] [PubMed]
17. Ambrogi, A.O. Biotic Invasions in a Mediterranean Lagoon. *Biol. Invasions* **2000**, *2*, 165–176. [CrossRef]
18. Goren, M.; Galil, B.S. A review of changes in the fish assemblages of Levantine inland and marine ecosystems following the introduction of non-native fishes. *J. Appl. Ichthyol.* **2005**, *21*, 364–370. [CrossRef]
19. Harmelin, M.; Bitar, G.; Harmelin, J.-G.; Monestiez, P. The littoral fish community of the Lebanese rocky coast (eastern Mediterranean Sea) with emphasis on Red Sea immigrants. *Biol. Invasions* **2005**, *7*, 625–637. [CrossRef]
20. Parravicini, V.; Azzurro, E.; Kulbicki, M.; Belmaker, J. Niche shift can impair the ability to predict invasion risk in the marine realm: An illustration using Mediterranean fish invaders. *Ecol. Lett.* **2015**, *18*, 246–253. [CrossRef]
21. Raitsos, D.; Beaugrand, G.; Georgopoulos, D.; Zenetos, A.; Pancucci, A.; Theocharis, A.; Papathanassiou, E. Global climate change amplifies the entry of tropical species into the Eastern Mediterranean Sea. *Limnol. Oceanogr.* **2010**, *55*, 1478–1484. [CrossRef]
22. Kalogirou, S.; Azzurro, E.; Bariche, M. The Ongoing Shift of Mediterranean Coastal Fish Assemblages and the Spread of Non-Indigenous Species. In *Biodiversity Enrichment in a Diverse World*; IntechOpen: London, UK, 2012.
23. Cinar, M.E.; Bilecenoglu, M.; Ozturk, B.; Katagan, T.; Yokes, M.B.; Aysel, V.; Dagli, E.; Acik, S.; Ozcan, T.; Erdogan, H. An updated review of alien species on the coasts of Turkey. *Mediterr. Mar. Sci.* **2011**, *12*, 257–315. [CrossRef]
24. Golani, D. The Sandy Shore of the Red Sea-Launching Pad for Lessepsian (Suez Canal) Migrant Fish Colonizers of the Eastern Mediterranean. *J. Biogeogr.* **1993**, *20*, 579–585. [CrossRef]
25. Galil, B.S.; Mienis, H.K.; Hoffman, R.; Goren, M. Non-indigenous species along the Israeli Mediterranean coast: Tally, policy, outlook. *Hydrobiologia* **2021**, *848*, 2011–2029. [CrossRef]
26. Por, F.D. *Lessepsian Migration: The Influx of Red Sea Biota into the Mediterranean by Way of the Suez Canal*; Springer: Berlin/Heidelberg, Germany, 1978. [CrossRef]
27. Stern, N.I.R.; Paz, G.U.Y.; Yudkovsky, Y.; Lubinevsky, H.; Rinkevich, B. The arrival of a second 'Lessepsian sprinter'? A first record of the red cornetfish *Fistularia petimba* in the Eastern Mediterranean. *Mediterr. Mar. Sci.* **2018**, *18*, 524–528. [CrossRef]
28. Ragheb, E. Morphometric and meristic characteristics of the first record *Fistularia petimba* (Lacepède, 1803) and Fistularia commersonii (Rüppell, 1838) (Piscès: Fistulariidae) from the Egyptian Mediterranean waters (West Alexandria). *Egypt. J. Aquat. Res.* **2022**, *48*, 143–150. [CrossRef]

29. Fritzsche, R.A. A Review of the Cornetfishes, Genus *Fistularia* (Fistulariidae), with a Discussion of Intrageneric Relationships and Zoogeography. *Bull. Mar. Sci.* **1976**, *26*, 106–204.
30. Fischer, W.; Bianchi, G.; Fisheries and Aquaculture Management Division. *FAO Species Identification Sheets for Fishery Purposes. Western Indian Ocean: Fishing Area 51*; Fischer, W., Bianchi, G., Eds.; FAO: Rome, Italy, 1984.
31. Sánchez-Tocino, L.; Hidalgo, F.; Pontes, M. Primera cita de Fistularia commersonii Ruppell, 1838 (Osteichtyes: Fistulariidae) en aguas mediterráneas de la Península Ibérica. *Zool. Baetica* **2007**, *18*, 79–84.
32. Kalogirou, S.; Corsini-Foka, M.; Kondylatos, G.; Wennhage, H. Diet of the invasive piscivorous fish Fistularia commersonii in a recently colonized area of eastern Mediterranean. *Biol. Invasions* **2007**, *9*, 887–896. [CrossRef]
33. Bañón, R.; Sande, C. First record of the red cornetfish *Fistularia petimba* (Syngnathiformes: Fistularidae) in Galician waters: A northernmost occurrence in the eastern Atlantic. *J. Appl. Ichthyol.* **2008**, *24*, 106–107. [CrossRef]
34. Cárdenas, S.; Alvarez-Berastegui, D.; Ortiz, J.M. First record of *Fistularia petimba* Laccepéde, 1803 (Pisces FIstulariidae) off the coast of Cadiz (southern Iberian Peninsula). *Boletín Inst. Español Oceanogr.* **1997**, *13*, 83–86.
35. Ünlüoğlu, A.; Akalın, S.; Dal, İ.; Tıraşın, E.; Aydın, C. First record of red cornetfish *Fistularia petimba* (Syngnathiformes: Fistulariidae) from Antalya and İskenderun Bays along Turkish coasts of the Mediterranean Sea. *J. Appl. Ichthyol.* **2018**, *34*, 977–980. [CrossRef]
36. Soydemir Çiftçi, N.; Ayas, D.; Doğangün, M. New locality record for *Fistularia petimba* Lacepède, 1803 from the Northeastern Mediterranean Sea (Mersin Bay). 2019. Available online: https://dergipark.org.tr/en/pub/medfar/issue/45471/491571 (accessed on 9 January 2024).
37. Karan, S.; Uyan, A.; Doğdu, S.; Gürlek, M.; Ergüden, D.; Turan, C. Genetic confirmation of Red cornetfish, *Fistularia petimba* (Syngnathiformes: Fistularidae) occurrence in Turkish marine waters. *FishTaxa* **2019**, *4*, 125–129.
38. Esri. ArcGIS Online | Cloud-Based GIS Mapping Platform. 2018. Available online: https://www.esri.com/en-us/arcgis/products/arcgis-online/overview (accessed on 16 April 2024).
39. Floerl, O.; Inglis, G.J. Starting the invasion pathway: The interaction between source populations and human transport vectors. *Biol. Invasions* **2005**, *7*, 589–606. [CrossRef]
40. Ferrario, J.; Caronni, S.; Occhipinti-Ambrogi, A.; Marchini, A. Role of commercial harbours and recreational marinas in the spread of non-indigenous fouling species. *Biofouling* **2017**, *33*, 651–660. [CrossRef]
41. Drake, J.M.; Lodge, D.M. Global hot spots of biological invasions: Evaluating options for ballast–water management. *Proc. R. Soc. London. Ser. B Biol. Sci.* **2004**, *271*, 575–580. [CrossRef] [PubMed]
42. Galil, B.S.; Boero, F.; Campbell, M.L.; Carlton, J.T.; Cook, E.; Fraschetti, S.; Gollasch, S.; Hewitt, C.L.; Jelmert, A.; Macpherson, E.; et al. 'Double trouble': The expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biol. Invasions* **2015**, *17*, 973–976. [CrossRef]
43. Ulman, A.; Ferrario, J.; Occhipinti, A.; Arvanitidis, C.; Bandi, A.; Bertolino, M.; Bogi, C.; Chatzigeorgiou, G.; Çiçek, B.; Içek, C.; et al. A massive update of non-indigenous species records in Mediterranean marinas. *PeerJ* **2017**, *5*, e3954. [CrossRef] [PubMed]
44. Katsanevakis, S.; Wallentinus, I.; Zenetos, A.; Leppäkoski, E.; Çinar, M.E.; Oztürk, B.; Grabowski, M.; Golani, D.; Cardoso, A.C. Impacts of invasive alien marine species on ecosystem services and biodiversity: A pan-European review. *Aquat. Invasions* **2014**, *9*, 391–423. [CrossRef]
45. Saad, A.A.; Sabour, W.G. Impact of invasive species on the biodiversity and fish stock; case study: *Fistularia commersonii* Ruppel, in the Syrian coast. *Rapp. Comm. Int. Mer Médit.* **2010**, *39*, 625.
46. Papageorgiou, M.; Resaikos, V.; Petrou, A. A preliminary assessment of *Fistularia petimba* (Lacepède, 1803) migration in the Mediterranean Sea: Historical and new data from Cyprus (Levantine Sea) with biological notes. *Mediterr. Mar. Sci.* **2023**, *24*, 446–453. [CrossRef]
47. Crocetta, F.; Al Mabruk, S.A.A.; Azzurro, E.; Bakiu, R.; Bariche, M.; Batjakas, I.E.; Bejaoui, T.; Ben Souissi, J.; Cauchi, J.; Corsini-Foka, M.; et al. "New Alien Mediterranean Biodiversity Records" (November 2021). *Mediterr. Mar. Sci.* **2021**, *22*, 724–746. [CrossRef]
48. Chirine, H.; Ibrahim, A.; Alshawy, F. First record of Red cornetfish, *Fistularia petimba* Lacepède, 1803 (Actinopterygii: Fistulariidae) from the Syrian coast. *Int. J. Aquat. Biol.* **2019**, *7*, 175–179. [CrossRef]
49. Ibrahim, A.; Alshawy, F.; Hussein, C. Confirmation records and new distribution of the red cornet fish *Fistularia petimba* lacepède, 1803 (actinopterygii: Fistulariidae) in the Syrian Marine Waters (Eastern Mediterranean). *Species* **2020**, *21*, 95–100.
50. Dragicevic, B.; Anadoli, O.; Angel, D.; Benabdi, M.; Bitar, G.; Castriota, L.; Crocetta, F.; Deidun, A.; Dulčić, J.; Edelist, D.; et al. New Mediterranean Biodiversity Records (December 2019). *Mediterr. Mar. Sci.* **2019**, *20*, 645–656. [CrossRef]
51. Cerim, H.; Yapıcı, S.; Gülşahin, A.; Soykan, O.; Bilge, G. The First Record of the Red Cornetfish (*Fistularia petimba* Lacepède, 1803) in the Aegean Sea. *Düzce Üniversitesi Bilim Teknol. Derg.* **2021**, *9*, 607–615. [CrossRef]

fishes

Communication

Population Dynamics Parameters and Exploitation Status of 55 Commercial Species in Egyptian Red Sea Fisheries: A Key to Sustainable Fisheries

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Abstract: Egyptian Red Sea fisheries face the same challenges as most of the world's fisheries, including overexploitation, habitat loss, IUU fishing, pollution, and climate change. These fisheries are highly diverse with multiple species targeted by multiple fleets, using different fishing gears. Much work has been performed in recent years to assess Red Sea fish stocks. However, not all fish stocks in the Egyptian Red Sea are assessed, and those that are assessed only cover 30% of landings. The assessments are unbalanced by area, with the Gulf of Suez being much better covered than the southern Red Sea and Gulf of Aqaba. The results show that most of the analyzed stocks are overexploited. There is an urgent need to take action to protect, conserve, and restore the different fish stocks in different fishing grounds. These actions will ensure the sustainability of the fisheries, making them ecologically friendly and economically and socially efficient.

Keywords: exploitation level; fisheries management; fish species; population parameters; Red Sea

Key Contribution: Population dynamics parameters are valuable for understanding fish populations and assessing stock health, enabling the development of sustainable management measures for commercially exploited species.

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1. Introduction

The Red Sea, a tropical sea with a rich history and unique marine ecosystem, is home to a wide variety of fish, many of which are of great commercial importance [1–3]. However, the unsustainable exploitation and illegal, unreported, and unregulated fishing (IUU) of these species threatens the health of the ecosystem and the long-term viability of the fishery [1,2,4]. Despite a growing demand for fishing resources due to population increase and a hard economic situation, sustainable management remains a major challenge for Egypt [5–7]. Understanding the biology of these species is essential to ensure the sustainability of fishing and the conservation of the marine ecosystem.

The Red Sea (Figure 1) is an elongated narrow sea between Northeastern Africa and the Arabian Peninsula, between 30° N and 12°30′ N and from 32° E to 43° E, with a straight-line length of about 2000 km and an average width of 208 km [5]. The Red Sea is connected to the Indian Ocean in the south through the narrow strait of Bab al Mandab. It has an average depth of 491 m, with a maximum depth of 2850 m. In the north, the Red Sea is divided into the Gulfs of Suez and Aqaba. The Red Sea is characterized by a few unique oceanographic and biological structures and is considered a hotspot for coral reef ecology. It also boasts high fish diversity, with more than 1400 species of fish reported in FishBase (www.fishbase.org, accessed on 1 April 2024).

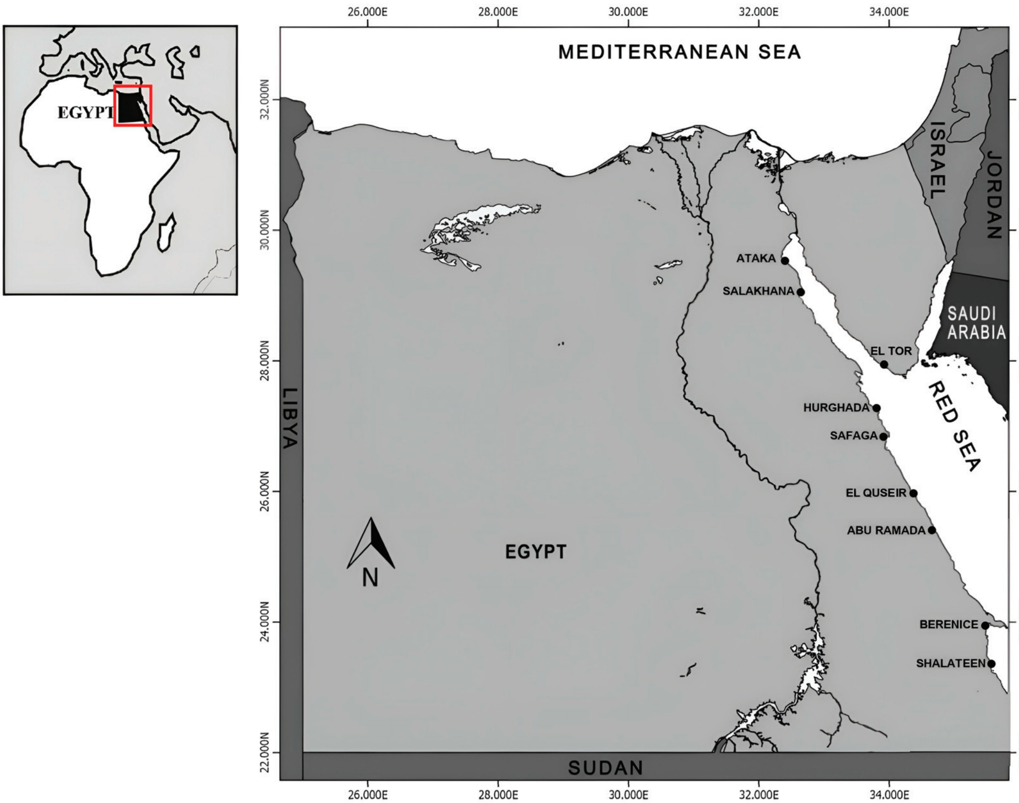

Figure 1. Egyptian Red Sea with the different fishing ports.

Three main fishing methods are deployed in Red Sea fisheries: bottom trawl, purse seine, and artisanal ones. Bottom trawls constitute about 20% of Egyptian Red Sea fisheries, forming about 54% of the gross revenue due to the high price of some demersals caught by it, such as shrimp and cephalopods. Purse seine constitutes about 68%, and the artisanal fishery contributes the remaining 12% of the Red Sea catch in Egypt [1].

This study aims to provide up-to-date scientific information on the biological parameters of more than 50 commercial fish species from the Red Sea, including data on catch, areas fished, as well as basic data on the biology and dynamics of these species, thereby contributing to sustainable fisheries management and the conservation of marine biodiversity in this unique region. In the absence of information in this regard, this contribution is the first attempt to compile population dynamics parameters and the exploitation status of 55 fish species in the Egyptian Red Sea. This information will provide a solid basis for assessing the current status of fish stocks and establishing reasonable catch limits.

2. Material and Methods

The Egyptian sector of the Red Sea (Figure 1) is about 1080 km long, extending from Suez City in the north to Mersa Halayeb in the south. Numerous fishing grounds are found along the Egyptian Red Sea, yielding an average annual catch of about 48 thousand tons [8]. The Egyptian Red Sea is divided into three main fishing grounds: the Gulf of Suez, the proper Red Sea (from Hurghada to Halayeb, including Foul Bay), and the Gulf of Aqaba. Several landing sites are located along the Egyptian Red Sea, including Ataka, El-Tor, Nuweiba, Hurghada, Safaga, Quseir, Berenice, Shalateen, and Abu Ramad. Additionally, Egypt has signed treaties with its neighbors to allow fishing in their territories (Figure 2).

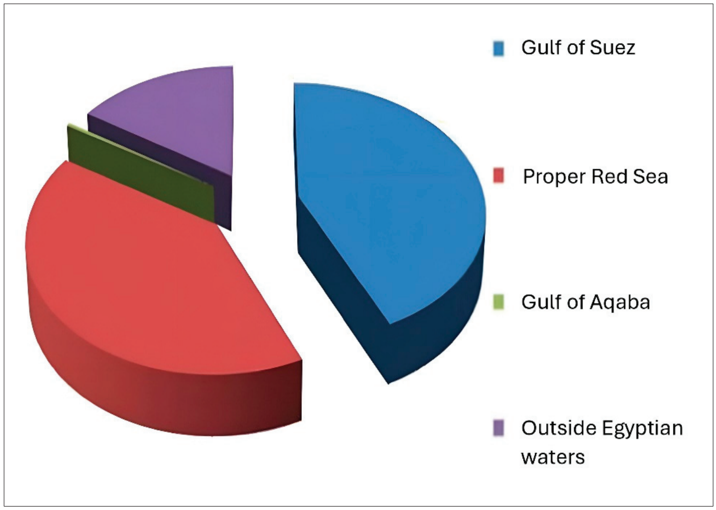

Figure 2. Contribution (%) of the main fishing grounds to the Red Sea's total catch.

The most common fishing gears used in the Red Sea by Egyptian fishermen are bottom trawls and purse seines (industrial fishery), hand lines, long lines, and gillnets (artisanal fishing), along with a variety of gears used by some of the traditional coastal communities. The industrial fishing fleets operate primarily in the Gulf of Suez and its neighboring areas, as well as Foul Bay. Semi-industrial fleets concentrate near Ataka, Salakhana, Hurghada, and El-Tor. Small-scale fisheries are common in Safaga, Quseir, Shalateen, and Abu Ramad [1].

This study relies on fishery statistics from the General Authority for Fish Resources Development (GAFRD) annual statistical book of 2021. Bimonthly field trips lasting at least seven days each have been carried out routinely since 1999 by the Fish Population Dynamics Lab, National Institute of Oceanography and Fisheries (NIOFs), to cover all fishing grounds along the Egyptian Red Sea. During these field trips, fish samples were collected directly from commercial fishing boats, and interviews were conducted with fishermen working in the area. This study analyzes data collected over the past ten years (2014–2024) in the Egyptian Red Sea with its two gulfs (Figure 1). This study focused on landings at the following locations: Ataka and El-Tor in the Gulf of Suez; Hurghada, Berenice, Shalateen, and Abu Ramad in the proper Red Sea; and Nuweiba and Dahab in the Gulf of Aqaba. In addition, the population parameters were compiled from two sources: previously published works and new estimations by the authors of this study for specific species.

Estimates of the asymptotic length ($L\infty$) and the growth rate (K) were obtained using the ELEFAN program (Electronic Length Frequency Analysis) within TropfishR [9] for R statistical program version 3.6.3 [10].

Analysis of the cumulative catch curve [11] and of the length-converted catch curve [12] was used to estimate the total mortality coefficient (Z).

Natural mortality (M) per age was estimated using an online tool (barefootecologist.com) with three empirical formulae. Fishing mortality (F) was then determined as $Z - M$.

The exploitation status (E) was calculated as $E = F/Z$, where F is the fishing mortality rate.

The exploitation rate (E) is crucial for evaluating a fish stock's status as optimal, underexploited, or overexploited. According to Gulland (1971), a stock is considered optimally fished when fishing mortality (F) equals natural mortality (M). In simpler terms, an exploitation rate (E) of 0.5 indicates optimal fishing pressure.

3. Results and Discussion

3.1. Landings and Catch Composition

Official annual landings of the Egyptian Red Sea fluctuated between 43.6 and 51.5 thousand tons from 2012 to 2021, averaging around 48 thousand tons. Red Sea fisheries account for about 11.5% of Egypt's total landed catch from natural resources, generating an estimated revenue of EGP 6.35 billion. Figures 2 and 3 illustrate the contribution of different fishing grounds to the Red Sea's total catch. Table 1 presents catch data from the GAFRD's 2021 statistical book. It is worth mentioning that the data recording system in Egypt is inaccurate. Many commercial species are not recorded separately but mixed in an "others" group. Also, there is no recorded data about bycatch and discards in the Egyptian marine fisheries. Therefore, urgent improvements are needed for the current recording system, as well as capacity-building programs to raise the qualifications of recorders at the different landing sites.

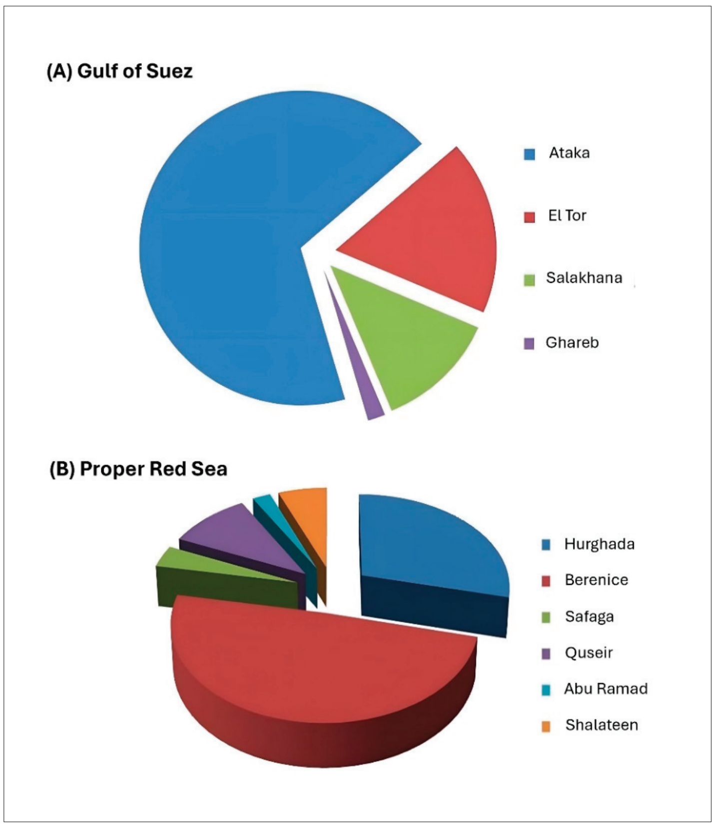

Figure 3. Contribution (%) of the main fishing ports to the Red Sea's total catch.

The field trips revealed a high diversity and abundance of fish species, especially in the Gulf of Aqaba, which the current catch statistics may not fully capture. Additionally, catch data are often recorded for groups of species rather than individual species.

The trawl fishery primarily catches fish from groups such as Synodontidae (lizardfish), Lutjanidae (snapper), Penaeidae (large shrimp), Mullidae (red mullet), Nemipteridae (breams), and Siganidae (rabbitfish). It is worth noting that large shrimp and cuttlefish are the most commercially valuable species in the trawl fishery, commanding high prices between EGP 300 and 500 per kilogram and representing about 45% of the trawl fishery's gross revenue.

Table 1. Available catch data based on GAFRD statistical book 2021 [8].

Family/Species	Total Catch (ton)	% of the Catch	By Area (ton)		
			Gulf of Suez	Gulf of Aqaba	Proper Red Sea
Lethrinidae	2652	5.86			
Lethrinus nebulosus	1914		205	14	1695
L. borbonicus	401		184	--	217
L. luntjan	--		--	--	--
L. mahsena	260		15	12	233
L. variegatus	77		77	--	--
L. microdon	--		--	--	--
L. miniata	--		--	--	--
Monotaxis grandoculis	--		--	--	--
Serranidae	3579	7.91			
Mix	2379		144	12	2223
Epinephelus summana	--		--	--	--
E. tauvina	31		7	6	18
E. chlorostigma	--		--	--	--
E. areolatus	87		87	--	--
Cephalopholis argus	--		--	--	--
C. oligostikta	--		--	--	--
C. miniata	--		--	--	--
Plectropomus maculatus	844		--	--	844
Variola louti	236		--	6	230
Plectropomus areolatus	2		2	--	--
Lutjanidae	777	1.72			
Lutjanus bohar	728	--	1	--	727
L. ehrenbergii	3	--	3	--	--
L. quinquelineatus	--	--	--	--	--
L. kasmira	--	--	--	--	--
Aphareus rutilans	46	--	--	--	46
Haemulidae	383	0.85	--	--	--
Pomadasys argenteus	34	--	--	--	34
P. stridens	315	--	211	--	104
Plectorhinchus shotaf	34	--	--	--	34
Synodontidae	2177	4.81	1417	--	760
Saurida undosquamis					
S. tumbil					
Mullidae	628	1.39	228	--	400
Mulloidichthys vanicolensis	--	--	--	--	--
M. flavolineatus	--	--	--	--	--
Upeneus vittatus	--	--	--	--	--
U. sulphureus	--	--	--	--	--
U. moluccensis	--	--	--	--	--
U. pori	--	--	--	--	--
Parupeneus forsskali	17	--	2	--	15
P. macronema	--	--	--	--	--
P. cyclostomus	--	--	--	--	--
Holocentridae	78	0.17	15	--	63
Sargocentron rubrum					
S. spiniferum					
Neoniphon sammara					
Carangidae	10,930	24.16			
Mix	8545		4984	--	3561
Trachurus indicus	--				
Decapterus macrosoma	--				
D. maruadsi	--				
Alepes djedaba	--				
Carangoides bajad	2385		5	--	2380

Table 1. *Cont.*

Family/Species	Total Catch (ton)	% of the Catch	By Area (ton)		
			Gulf of Suez	Gulf of Aqaba	Proper Red Sea
Nemipteridae	1321	2.92	373	--	948
Nemipterus japonicas					
N. zysron					
N. randalli					
Sparidae	334	0.73			
Rhabdosargus haffara	--				
Acanthopagrus bifasciatus	--				
Argyrops spinifer	310		90	2	218
Diplodus nokt	24		24	--	--
Gerridae	132	0.29	20	3	109
Gerres oyena					
Scaridae	980	2.17	165	9	806
Hipposcarus harid					
Chlorurs sordidus					
Siganidae	377	0.83			
Siganus rivulatus	377		164	--	213
Scombridae	3009	6.65			
Scomber japonicas	73		73	--	--
Rastrelliger kanagurta	2369		150	--	2219
Euthynnus affinis	567		66	--	501
Clupeidae	10,641	23.52			
Sardinella aurita	8297		44	--	8253
Etrumeus teres	1120		1120	--	--
Small sardine *	1224		1224	--	--
Engraulidae	4000	8.84	4000	--	
Penaeidae	266	0.59	159	--	107
P. semisulcatus				--	
P. japonicas				--	
Sepiidae	111	0.45	102	--	9
Sepia pharaoni	--				
S. prashadi	--				
S. dollfusi	--				
Portunidae	188	0.41	156		32
Portunus pelagicus					
Loliginidae	507	1.12			
Loligo spp.			140	--	367

* Small sardine composed of small-sized sardines such as *Dussumieria acuta*, *Amblygaster sirm*, *Amblygaster clupeoides*, *Ilisha melastoma*, and *Herklosichthys punctatus* (personal observation). -- indicates an absence of data in the national recording system.

While the dominant fish families in the artisanal catch include groupers (Serranidae), emperors (Lethrinidae), and longspine bream (Sparidae), other Scombridae (little tuna and Spanish mackerel) are also present. More than 100 fish species from up to 20 families, like needlefish, squirrelfish, goatfish, and rabbitfish, were grouped together in the “others” category without any sorting or identification [1,13,14]. This “others” group constitutes about 46.5 percent of the total artisanal landings, highlighting the lack of accuracy and detail in the current fishery statistics collection and recording system.

Unlike the Gulf of Suez, where industrial fisheries dominate, artisanal fishing is the primary activity in the proper Red Sea and Gulf of Aqaba. It supplies approximately 40% of the total fish production from the proper Red Sea and constitutes the total of the Gulf of Aqaba’s catch [8,13–19].

3.2. Population Parameters and Exploitation Level

As shown in Table 2, the population parameters and status of 55 fish, crustacean, and mollusk species from the Egyptian Red Sea are presented. Despite the fact that both of Mehanna's (2001) [20,21] studies might be outdated, they are included in Table 2 to capture all the available data. Mehanna's (2001) [20] study is the only one that has assessed *Rhabdosargus haffara* in the Gulf of Suez, although Osman et al. (2020) [22] more recently evaluated the same species in the Red Sea off the Hurghada fishing area. Similarly, Mehanna's (2001) [21] study remains the sole one on *Rastrelliger kanagurta* in the Gulf of Suez. Based on Gulland's (1971) interpretation of the exploitation level, only five fish species are maintained at below slightly optimum exploitation levels: *Lethrinus borbonicus*, *L. variegatus*, and *Euthynnus affinis* from the Gulf of Aqaba and *Monotaxis grandoculis* and *Parupeneus forsskali* from Hurghada in the proper Red Sea. The remaining species, 50 out of the 55 assessed commercial species (90.9%), were found to be overexploited.

Table 2. Growth parameters and mortality estimates of fish species from Red Sea, Egypt.

Species	Area	Author	Parameters					
			L∞	K	F	M	E	
Lethrinus nebulosus	Gulf of Suez	Present study	71.4	0.24	0.55	0.29	0.65	
	Hurghada	Present study	73.6	0.22	0.61	0.29	0.68	
L. borbonicus	Gulf of Suez	Present study	32.5	0.42	0.81	0.45	0.64	
	Aqaba Gulf	Mehanna, 2023 [17]	35.8	0.48	0.70	0.72	0.49	opt
	Shalateen	Mehanna, 2011 [16]	34.57	0.53	1.65	0.62	0.73	
L. luntjan	Red Sea	Zaahkouk et al., 2017 [23]	56.95	0.28	1.17	0.35	0.77	
L. mahsena	Gulf of Suez	Present study	61.8	0.29	1.05	0.33	0.76	
L. variegatus	Aqaba Gulf	Mehanna et al., 2024 [23]	25.79	0.51	0.80	1.01	0.44	opt
	Hurghada	Present study	26.11	0.50	1.11	0.67	0.62	
L. microdon	Red Sea	Mehanna et al., 2017 [24]	67.46	0.25	0.74	0.30	0.71	
Monotaxis grandoculis	Hurghada	ElMahdy et al., 2022 [25]	53.87	0.23	0.46	0.52	0.47	opt
Epinephelus summana	Hurghada	Mehanna et al., 2022 [26]	63.39	0.13	0.49	0.33	0.60	
E. areolatus	Shalateen	Mehanna, 2005 [27]	48.79	0.31	0.55	0.13	0.81	
E. tauvina	Hurghada	Present study	106.3	0.11	0.79	0.22	0.78	
Cephalopholis argus	Hurghada	Mehanna et al., 2019 [26]	44.22	0.26	0.74	0.56	0.57	
C. miniata	Quseir	Mohammad, 2007 [28]	39.99	0.23	1.26	0.58	0.68	
Variola louti	Quseir	Mohammad, 2007 [28]	62.57	0.14	0.63	0.37	0.63	
L. ehrenbergii	Hurghada	Mehanna et al., 2017 [29]	32.4	0.47	1.48	0.69	0.68	
L. quinquelineatus	Hurghada	Mehanna et al., 2017 [29]	35.5	0.35	1.38	0.61	0.69	
L. kasmira	Shalateen	Baker&Mehanna, 2024 [30]	33.76	0.35	0.99	0.59	0.62	
P. stridens	Gulf of Suez	Mehanna et al., 2023 [31]	21.7	0.40	1.76	0.74	0.70	
Saurida undosquamis	Baraneis	Mehanna, 2022 [32]	48.66	0.33	1.28	0.45	0.74	
S. tumbil	Baraneis	Mehanna, 2022 [32]	35.30	0.43	1.19	0.46	0.72	
Mulloidichthys vanicolensis	Hurghada	Farrag et al., 2018 [33]	33.30	0.37	0.79	0.64	0.56	
M. flavolineatus	Hurghada	Farrag et al., 2018 [33]	38.0	0.27	1.01	0.48	0.68	
Upeneus moluccensis	Gulf of Suez	Present study	22.99	0.38	1.06	0.48	0.69	
U. pori	Gulf of Suez	Present study	20.85	0.40	0.85	0.44	0.66	
Parupeneus forsskali	Hurghada	Farrag et al., 2018 [33]	31.62	0.32	0.54	0.55	0.49	opt
Sargocentron spiniferum	Shalateen	Mohammad et al., 2020 [19]	53.25	0.23	0.57	0.47	0.55	
Neoniphon sammara	Hurghada	ElMahdy et al., 2023 [34]	26.49	0.28	0.82	0.57	0.59	
Trachurus indicus	Gulf of Suez	Present study	24.81	0.51	0.91	0.36	0.72	
Decapterus macrosoma	Gulf of Suez	Present study	25.11	0.56	0.90	0.32	0.74	
D. maruadsi	Gulf of Suez	Present study	26.21	0.51	1.06	0.56	0.65	
Carangoides bajad	Shalateen	Mohammad et al., 2022 [35]	57.69	0.24	1.52	0.34	0.82	
Caranx melampygus	Shalateen	Mohammad et al., 2022 [35]	70.11	0.17	1.75	0.28	0.86	
Alepes djedaba	Hurghada	Present study	41.12	0.41	1.21	0.46	0.72	
Nemipterus japonicas	Gulf of Suez	Present study	34.15	0.33	1.18	0.56	0.68	
N. zysron	Aqaba Gulf	Present study	29.31	0.51	0.90	0.62	0.59	
N. randalli	Hurghada	Present study	28.11	0.54	1.38	0.66	0.68	
Rhabdosargus haffara	Gulf of Suez	Mehanna, 2001 [20]	26.79	0.47	0.90	0.30	0.76	
	Hurghada	Osman et al., 2020 [22]	30.47	0.36	1.32	0.66	0.67	
Acanthopagrus bifasciatus	Hurghada	ElMahdy et al., 2019 [36]	65.62	0.21	1.53	0.46	0.77	
Gerres oyena	Gulf of Suez	Present study	26.11	0.38	0.81	0.44	0.65	
Hipposcarus harid	Hurghada	Osman, 2015 [37]	57.16	0.23	1.75	0.50	0.78	
Chlorurs sordidus	Hurghada	Osman, 2015 [37]	40.27	0.28	0.82	0.60	0.58	
Siganus rivulatus	Gulf of Suez	Present study	25.31	0.44	1.72	0.67	0.72	
	Shalateen	Present study	36.44	0.36	1.02	0.37	0.73	

Table 2. *Cont.*

Species	Area	Author	Parameters					
			L∞	K	F	M	E	
Scomber japonicas	Gulf of Suez	Mehanna, 2002 [38]	33.06	0.48	1.82	0.64	0.74	
Rastrelliger kanagurta	Gulf of Suez	Mehanna, 2001 [21]	32.15	0.57	0.82	0.26	0.76	
Euthynnus affinis	Aqaba Gulf	Mehanna, 2024 [18]	83.64	0.47	0.65	0.77	0.46	opt
Sardinella aurita	Gulf of Suez	Present study	23.33	0.48	1.05	0.27	0.79	
S. gibbosa	Gulf of Suez	Present study	33.15	0.47	0.89	0.57	0.60	
Etrumeus teres	Red Sea	Mehanna and Elgammal (2005) [39]	26.97	0.59	1.46	0.87	0.63	
Engraulis japonicas	Gulf of Suez	Present study	15.45	0.61	0.71	0.34	0.68	
E. encrasicolus	Gulf of Suez	Present study	12.61	0.55	1.49	1.02	0.59	
Penaeus semisulcatus	Gulf of Suez	Mehanna, 2000 [40]	26.61	1.69	5.45	2.33	0.70	
P. japonicas	Gulf of Suez	Present study	24.3	1.71	4.73	2.45	0.66	
Sepia pharaoni	Gulf of Suez	Mehanna et al., 2009 [15]	23.48 ML	0.59	2.11	0.77	0.73	
S. savignyi	Gulf of Suez	Mehanna and Elgammal (2010) [41]	31.29 ML	0.54	0.89	0.57	0.60	
S. dollfusi	Gulf of Suez	Mehanna and Amin (2005) [42]	16.9 ML	0.91	2.50	1.07	0.70	
Portunus pelagicus	Gulf of Suez	Present study	10.96 CL	1.44	6.56	2.11	0.76	

ML: Mantle Length, CL: Carapace Length.

4. Conclusions

Our analysis of the Egyptian Red Sea's most commercially important fish species reveals a high fishing mortality, which indicates a high exploitation level. Therefore, a reduction in exploitation rate and consequently the fishing mortality by about 20 to 70% is suggested to reach the optimum level (E_{opt}), which would help ensure the sustainability of the fish stocks. This can be achieved through enforcement of the closed fishing season and enforcement of the minimum legal size to conserve the spawning stock. The basic biological information generated from this study will be valuable for further population studies and stock assessment. These findings, in turn, can be applied to develop sustainable management measures for these commercially exploited fish species.

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References

1. Mehanna, S.F. Egyptian Marine Fisheries and its sustainability. In *Sustainable Fish Production and Processing*; Galanakis, C.M., Ed.; Academic Press: Cambridge, MA, USA; Elsevier: Amsterdam, The Netherlands, 2021; pp. 111–140.
2. Samy-Kamal, M.; Mehanna, S.F. Evolution of fishing effort and fishing capacity during the last two decades (2000–2019) in Egypt's marine fisheries: Spotting the fleet overcapacity. *Reg. Environ. Chang.* **2023**, *23*, 113. [CrossRef]
3. Samy, M.; Lizaso, J.S.; Forcada, A. Status of marine protected areas in Egypt. *Biodivers. Conserv.* **2011**, *34*, 165–177. [CrossRef]
4. Samy-Kamal, M. Insights on illegal, unreported and unregulated (IUU) fishing activities by Egyptian vessels in neighbouring countries. *Fishes* **2022**, *7*, 288. [CrossRef]
5. Samy-Kamal, M. Status of fisheries in Egypt: Reflections on past trends and management challenges. *Rev. Fish Biol. Fish.* **2015**, *25*, 631–649. [CrossRef]
6. Samy-Kamal, M. Outlook on the fisheries policy reform in Egypt and the draft of the new fisheries law. *Mar. Policy* **2020**, *120*, 104136. [CrossRef]
7. Samy-Kamal, M. Prices in the Egyptian seafood market: Insights for fisheries management and food security. *Fish. Res.* **2021**, *233*, 105764. [CrossRef]
8. *GAFRD Annual Reports 2012–2021*; GAFRD Fisheries Statistics Yearbook; General Authority for Fish Resources Development: Cairo, Egypt, 2022.

9. Mildenberger, T.K.; Taylor, M.H.; Wolff, M. TropFishR: An R package for fisheries analysis with length-frequency data. *Methods Ecol. Evol.* **2017**, *8*, 1520–1527. [CrossRef]
10. R Core Team. *R A language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2020; Available online: https://www.R-project.org/ (accessed on 1 April 2024).
11. Jones, R.; van Zalinge, N.P. Estimates of mortality rate and population size for shrimp in Kuwait waters. *Kuwait Bull. Mar. Sci.* **1981**, *2*, 273–288.
12. Pauly, D. Length converted catch curves: A powerful tool for fisheries research in the tropics (Part 1). *Fishbyte* **1983**, *1*, 9–13.
13. Mehanna, S.F. An assessment and management of the coral reef fish stocks in the Gulf of Suez. *Egypt. J. Aquat. Biol. Fish.* **1999**, *3*, 103–114.
14. Mehanna, S.F.; El-Gammal, F.I. Gulf of Suez fisheries: Current status, assessment and management. *J. King Abdulaziz Univ. Mar. Sci.* **2007**, *18*, 3–18. [CrossRef]
15. Mehanna, S.F.; Hegazi, M.M.; El-Sherbeny, A.S. Stock assessment and management of the cuttlefish *Sepia pharaoni* (Mollusca: Cephalopoda) in the Gulf of Suez. *Egypt. J. Aquat. Biol. Fish.* **2009**, *13*, 421–431.
16. Mehanna, S.F. Population dynamics and management of snubnose emperor *Lethrinus bungus* (*L. borbonicus*) from the Foul Bay, Red Sea. In Proceedings of the INOC-XI International Symposium, Bogor, Indonesia, 25–27 October 2011; pp. 121–129.
17. Mehanna, S.F. Population Dynamics and Assessment of the Snubnose Emperor *Lethrinus borbonicus* (*L. bungus*) from the Gulf of Aqaba, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2023**, *27*, 643–654. [CrossRef]
18. Mehanna, S.F. Life history parameters and stock status of kawakawa, *Euthynnus affinis* (Cantor, 1849) from the Gulf of Aqaba, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2024**, *28*, 299–312. [CrossRef]
19. Mohammad, A.S.; Mehanna, S.F.; Osman, Y.A.; El-Mahdy, S.M. Age, growth and population parameters of the spiny squirrelfish, *Sargocentron spiniferum* (Forsskål, 1775) from Shalateen fishing area, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2020**, *24*, 469–480. [CrossRef]
20. Mehanna, S.F. Growth, mortality and yield per recruit of *Rhabdosargus haffara* (Sparidae) from the Suez Bay. *Egypt. J. Aquat. Biol. Fish.* **2001**, *5*, 31–46. [CrossRef]
21. Mehanna, S.F. Dynamics and management of the Indian mackerel *Rastrelliger kanagurta* (Cuvier, 1816) in the Gulf of Suez, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2001**, *5*, 179–194. [CrossRef]
22. Osman, Y.A.; Mehanna, S.F.; El-Mahdy, S.M.; Mohammad, A.S.; Mahe, K. Age precision and growth rate of *Rhabdosargus haffara* (Forsskål, 1775) from Hurghada fishing area, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2020**, *24*, 341–352. [CrossRef]
23. Zaahkouk, S.A.; Khalaf-Allah, H.M.; Mehanna, S.F.; El-Gammal, F.I.; Makkey, A.F. Studies on age, growth, and mortality rates for management of the redspot emperor, *Lethrinus lentjan* (Lacepède, 1802) in the Egyptian sector of Red Sea. *Egypt. J. Aquat. Biol. Fish.* **2017**, *21*, 63–72. [CrossRef]
24. Mehanna, S.F.; El-Gammal, F.I.; Zaahkouk, S.A.; Khalaf-Allah, H.M.; Makkey, A.F. Population dynamics of the small tooth emperor, *Lethrinus microdon* (Valenciennes, 1830) from the Egyptian Red Sea. *Int. J. Fish. Aquat. Stud.* **2017**, *5*, 158–163.
25. El-Mahdy, S.M.; Mehanna, S.F.; Mohammad, A.S.; Osman, Y.A. Age, growth, mortality and yield per recruit of humpnose big-eye bream *Monotaxis grandoculis* (Forsskål, 1775) from Hurghada, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2022**, *26*, 857–868. [CrossRef]
26. Mehanna, S.F.; Osman, Y.A.A.; Khalil, M.T.; Hassan, A. Age and growth, mortality and exploitation ratio of *Epinephelus summana* (Forsskål, 1775) and Cephalopholis argus (Schneider, 1801) from the Egyptian Red Sea coast, Hurghada fishing area. *Egypt. J. Aquat. Biol. Fish.* **2019**, *23*, 65–75. [CrossRef]
27. Mehanna, S.F. Population dynamics of the areolate grouper Epinephelus areolatus from the Egyptian sector of Red Sea. In Proceedings of the 12th International Conference of Union of Arab Biologists, El-Hodeida University, Al Hudaydah, Yemen, 19–24 November 2005.
28. Mohammad, A.S. Population Dynamics and Stock Assessment of Some Species of Genus Cephalopholis and Genus Variola from the Red Sea, Egypt. MSc Thesis, Assiut University, Asyut, Egypt, 2007; 178p.
29. Mehanna, S.F.; Soliman, F.M.; Soliman, H.A.; Baker, T.S. Age, growth and length-weight relationship of *Lutjanus quinqelineatus* (Boch, 1790) and Lutjanus ehrenbergii (Petere, 1869) from the Red Sea, Egypt. In Proceedings of the Al-Azhar University, Assiut Branch Annual Conference, Hurghada, Egypt, 5–7 August 2017.
30. Baker, T.S.; Mehanna, S.F. Some biological aspects and life history parameters of common bluestripe snapper *Lutjanus kasmira* (Family: Lutjanidae) from Shalatein, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2024**, *28*, 411–422. [CrossRef]
31. Mehanna, S.F.; Ali, M.M.; Abdella, A.N.E. Evaluation of fishery status and some biological aspects of striped piggy, *Pomadasys stridens* from Gulf of Suez, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2023**, *27*, 791–803. [CrossRef]
32. Mehanna, S.F. Population dynamics and stock assessment of two Lizardfish species in the southern Red Sea coasts, Foul Bay, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2022**, *26*, 299–311. [CrossRef]
33. Farrag, M.M.; Osman, A.G.M.; Mehanna, S.F.; Osman, Y.A. Fisheries status of the common species of family Mullidae in the Southern Red Sea, Hurghada, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2018**, *22*, 249–265.
34. El-Mahdy, S.M.; Mohammad, A.S.; Osman, Y.A.; Hassanein, E.M.; Mehanna, S.F. Age, Growth, Mortality and exploitation level of Squirrelfish, *Neoniphon sammara* (Forsskal, 1775) from Hurghada, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2023**, *27*, 51–63. [CrossRef]

35. Mohammad, A.S.; Mehanna, S.F.; Mahmoud, U.M. Age and growth based on the scale readings of the two carangid species *Carangoides bajad* and *Caranx melampygus* from Shalateen fishing area, Red Sea, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2022**, *26*, 399–411. [CrossRef]
36. El-Mahdy, S.M.; Mehanna, S.F.; Mahmoud, U.M.; El-Gammal, F.I. Population Dynamics and Management of Two-barred Seabream Acanthopagrus Bifasciatus in the Red Sea, Egypt. *Int. J. Ecotoxicol. Ecobiol.* **2019**, *4*, 80–87.
37. Osman, Y.A. Population dynamics and stock assessment of the commercial species of family Scaridae in the Southern Red Sea, Egypt. MSc Thesis, Port Said University, Port Fuad, Egypt, 2015.
38. Mehanna, S.F. Fisheries management of the slimy mackerel Scomber japonicus in the Gulf of Suez based on relative yield per recruit analysis. *Egypt. J. Aquat. Biol. Fish.* **2002**, *6*, 217–232. [CrossRef]
39. Mehanna, S.F.; El-Gammal, F.I. Stock assessment of the round herring, *Etrumeus teres* (DeKay, 1842) from the Egyptian sector of Red Sea. *Indian J. Fish.* **2005**, *52*, 377–383.
40. Mehanna, S.F. Population dynamics of Penaeus semisulcatus in the Gulf of Suez, Egypt. *Asian J. Fish.* **2000**, *13*, 127–137. [CrossRef]
41. Mehanna, S.F.; El-Gammal, F.I. Growth and population dynamics of cuttlefish *Sepia savignyi* in the Gulf of Suez, Red Sea. *Indian J. Fish.* **2010**, *57*, 1–6.
42. Mehanna, S.F.; Amin, A.M. Population dynamics of the cuttlefish *Sepia dollfusi* (Adam, 1941) from the Gulf of Suez, Egypt. *Egypt. J. Aquat. Biol. Fish.* **2005**, *9*, 1–14. [CrossRef]

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Technical Efficiency of the Nile Perch Fishing Fleet on Lake Victoria: A Comparative Perspective on the Three Riparian Countries Kenya, Tanzania and Uganda

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Abstract: Lake Victoria, which is shared by Kenya, Tanzania, and Uganda, faces escalating concerns over sustainable fisheries amidst expanding fishing efforts. This study aims to investigate how technical efficiency (TE) and labor productivity (LP) of the Nile perch fishing fleet vary across the three riparian countries. Using a nine-year dataset spanning from 2005 to 2021 and employing Stochastic Frontier Analysis, this study evaluates the TE of the fleet, where LP is determined as catch per crew hour fished in a day for three vessel types: motorized, paddled, and sailed. Motorized fleets had the highest mean technical efficiency (0.60–0.66), compared to paddled (0.29–0.49), and sailed vessel categories (0.24–0.46). Sailed vessels declined in all countries owing to their low TE. In Kenya, TE and LP increased for paddled vessels, especially in the period from 2015 to 2021, and a slight increase was also indicated for motorized vessels. Conversely, Uganda and Tanzania experienced gradual declines in TE and LP, particularly from 2015 to 2021, a period of rigorous law enforcement that led to declines in the number of paddled vessels by 50% and 7%, respectively, and a contrasting increase in motorized vessels. By 2021, the number of Ugandan motorized vessels had increased greatly but TE had declined compared to Kenya and Tanzania, a sign of overcapacity. The findings underscore the need for region-specific policies that address economic differences, policy implementation impacts, and resource health to promote sustainable transboundary fisheries management on Lake Victoria.

Keywords: fisheries management; fisheries technical efficiency; labor productivity; catch assessment; over capacity

Key Contribution: This study presents a novel cross-border comparison of technical efficiency and labor productivity in Lake Victoria's fishing fleet. Differences in vessel performance and labor output are shaped by factors such as economic growth, technology use, and regulations, with motorization emerging as a key driver of improved fleet efficiency.

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1. Introduction

Fisheries on Lake Victoria, Africa's largest freshwater lake, are an important socio-economic activity. The fisheries contribute significantly to regional food security and provide employment and livelihoods for the large lakeside populations [1]. The fisheries sector is of major economic importance and integral to the economies of the three riparian countries, contributing approximately 0.8%, 1.7%, and 3% of the Gross Domestic Product (GDP) in Kenya, Tanzania, and Uganda, respectively [2]. Lake Victoria's fisheries operate under limited restrictions on access. The three countries of Kenya (6%), Tanzania (51%), and Uganda (43%), which share the transboundary lake, have imposed regulations on fishing gear, registrations, and type of vessels to regulate fisheries efforts, but allow entry to the fishery after payment of a nominal access fee [3]. Fisheries management efforts suffer from access to reliable data and are plagued by data scarcity in both spatial and temporal dimensions, as well as irregular data collection practices. It is therefore important to seek ways in which the existing although limited data can be used [4].

The evolution of landed fish catches in Lake Victoria reflects notable shifts toward the focus on the Nile perch in the 1990s, which remains the most valued species and the primary fish export for the past three decades [5,6]. The significance of the Nile perch fishery is further underscored by the distribution of fishing effort, with up to 58% of the 210,620 fishers targeting the species, along with a comparable proportion of the 70,995 fishing crafts [7,8].

On the lake, fishery-related technological changes introduced by the early colonial governments replaced the inefficient and ancient traditional fishing methods. Modern fishing equipment, including synthetic gill nets and trawls, were used to increase catches per input, and outboard engines were introduced to expand access to fishing grounds [5,9]. The commercial importance of capture fisheries grew alongside increased markets and infrastructure development, leading to increased fishing efforts. This evolution has led to a shift in efficiency.

Technological advancements have led to reduced costs and transformed fishing fleets' performance in Lake Victoria. Three main types of vessel propulsion are used on the lake: motorized vessels with outboard engines, paddled vessels, and sail-powered vessels. The introduction of outboard engines in the 1950s led to sizeable changes in efficiency [9,10]. However, investment capacity is limited, and a small section of sailed vessels remains. The final vessel type, paddled vessels, is generally smaller than the other two, and its activities are limited to areas close to the shoreline [11].

While comparative studies on fleet performance have been conducted for some of the African lakes [12,13], no comparative study has been undertaken to assess the technical efficiency of the fishing fleets across the three riparian countries sharing Lake Victoria. Previous studies on technical efficiency have been conducted in individual countries, including research by [14] in Uganda and studies by [15,16] in Tanzania using cross-sectional data. In contrast, this study extends its analysis to nine years of panel data collected over 17 years (2005–2008; 2010–2011; 2014–2015; 2021). The utilization of panel data provides a unique opportunity to capture the dynamic and heterogeneous nature of fleet production units over time, considering factors such as country-specific technology adoption and economic and policy changes that may influence fleet technical efficiency. In addition to evaluating technical efficiency, the study also provides estimates of labor productivity (LP) for each vessel type across the three countries. LP, defined as the output (fish catches) per fisher over a specific period, is important in understanding the development of the fishery, given that Lake Victoria's fishery remains labor-intensive [17–19].

This study's objective is to assess the technical efficiency (TE) and labor productivity (LP) of the fishing fleet on Lake Victoria, comparing performance across Kenya, Tanzania, and Uganda, while identifying the key factors influencing these metrics.

The guiding research objectives are:

- To evaluate the status and historical development of the fishing fleet on Lake Victoria across the three riparian countries.
- To analyze the TE and LP scores of the fishing fleet across Kenya, Tanzania, and Uganda.
- Assess the impact of fleet development, regulatory frameworks, and fish stock health on TE and LP scores, and their influence on sustainability in Lake Victoria fisheries.

2. Materials and Methods

2.1. Measuring Technical Efficiency

Measurement of productive efficiency is commonly applied to fisheries to evaluate outcomes, policies, and development [20–24]. The Stochastic Frontier Approach (SFA) is used to estimate the technical efficiency of the Nile perch fleet. It models the relationship between outputs, such as fish catch, and inputs, such as fuel and labor, using a flexible functional form that represents underlying technology. The model is well-suited for single-species fisheries with multiple inputs and a single output, such as the Nile perch catches evaluated in this study [20,21], is flexible in dealing with complexity, and is versatile with respect to analyzing external factors of inefficiency [20,25,26]. The general approach is discussed in [27,28]. The model contains a composite error term, a random deviation, and an inefficiency term. The inefficiency term can contain a model of explanatory variables linking independent variables to the level of inefficiency [29]. Figure 1 shows a representation of the model terms.

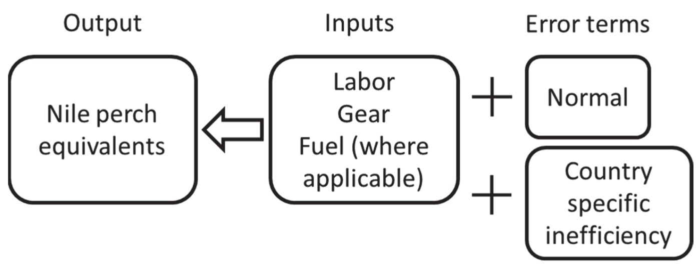

Figure 1. Representation of the empirical model used in the analysis, with output, inputs, and a composite error term.

2.2. Data Sources and Treatment

Two main datasets were used, a frame survey (FS) and catch assessment (CAS), with both obtained from the Lake Victoria Fisheries Organisation (LVFO) database. Frame survey data are generated from a complete census of all fishery variables, including the vessels, gears, and landing site facilities along the lake [7]. The data collected biennially on Lake Victoria was available for the period from 2000 to 2020 and was specifically used to answer this study's first objective, namely, to evaluate the status and historical development of the fishing fleet on Lake Victoria, including trends from 2000 to 2021 across the three riparian countries.

The second dataset and main dataset used to estimate TE and LP comprised the CAS data. It consists of nine-year vessel-level catch data (series of catch assessment surveys conducted with support from the Implementation of Fisheries Management Plan (IFMP) project during 2005–2008; and Lake Victoria Environmental Management Program (LVEMP1) 2011 and 2014 and 2015 by LVEMP2.) (2005–2008; 2010–2011; 2014–2015; 2021), collected over 17 years. The LVFO periodic survey data usually follows a two-stage sampling procedure where 10% of the landing sites in each country are identified as strata in the first stage and then vessels are randomly sampled at the landing sites in the second stage [30,31]. To address the missing variable of fuel use for motorized vessels, a supplemental survey was conducted: in Uganda between June and August 2017, and in Kenya and Tanzania from April to September 2020. Data were collected following the CAS data collection form, including vessel fuel use in liters as a variable. Fuel is a crucial input, especially for motorized vessels, as it is used to power engines and enable vessels to access their desired fishing grounds. The data obtained from the survey was used to predict fuel use for nine of the years in the period between 2005 and 2021. Details of the model are provided in Supplementary Materials.

The panel data, which consists of repeated observations of the same subjects over time [32], was organized as a series of independent cross-section surveys conducted between 2005 and 2021. Observations were grouped based on vessel propulsion as paddled, motorized, or sailed using gillnets and longlines and harvesting Nile perch. Initially, the CAS datasets were assessed independently to understand their structure, variables, coding, and measurements across different years. To ensure consistency throughout the nine years of sampling, data variables were renamed and re-coded wherever necessary, specifically to consolidate changes made in the standard operating procedures used for data collection in 2021 [30,31].

2.3. Variable Selection

Inputs included in the model were the number of units of gear, fuel (liters per fishing trip, where applicable), and labor (crew hours per trip), with the catch as the output variable. A single output measure (Nile perch quantity) was used for consistency [33,34]. In cases where bycatch such as Nile tilapia was present in the catch, the output was standardized to a Nile perch equivalent by dividing the catch value by the price of Nile perch.

2.4. Labor Productivity Computation

Labor productivity was calculated as the ratio of total fish catches (standardized to Nile perch equivalents) to the total labor input (measured as a product of the number of fishing crew in a vessel and hours fished in a day—24 h) for each vessel type [17]. The analysis was conducted separately for each country to identify differences in LP across the riparian states.

2.5. Data Summary Statistics

Table 1 provides a comprehensive overview of essential statistics for the output and input variables examined in this study. Sampled motorized vessels were highest in Uganda (50.8%), paddled vessels in Tanzania (52.1%), and sailed vessels dominated (45.9%) in Kenya.

Table 1. Summary statistics for the SFA model variables for the different vessel groups.

	Vessels			
	Motorized (N = 30,052)	**Paddled (N = 26,147)**	**Sailed (N = 19,192)**	**Total (N = 75,391)**
Country				
Kenya	2375 (7.9%)	1321 (5.1%)	8809 (45.9%)	12,505 (16.6%)
Tanzania	12,417 (41.3%)	13,631 (52.1%)	8252 (43.0%)	34,300 (45.5%)
Uganda	15,260 (50.8%)	11,195 (42.8%)	2131 (11.1%)	28,586 (37.9%)
Vessels by gear type				
GN	25,122 (83.6%)	14,026 (53.6%)	7659 (39.9%)	46,807 (62.1%)
LL	4930 (16.4%)	12,121 (46.4%)	11,533 (60.1%)	28,584 (37.9%)
Gear units				
Gillnets	61.373(20.058)	35.719 (24.877)	47.603(23.559)	51.433 (24.932)
Long lines	951.140 (740.370)	580.912 (559.374)	791.651 (452.081)	729.800(573.630)
Catch				
Mean (SD)	32.286 (37.101)	23.426 (25.704)	24.484 (29.592)	27.227 (31.904)
Range	0.000–705.000	0.000–470.000	0.000–1000.000	0.000–1000.000
Labor				
Mean (SD)	27.333 (14.231)	28.740 (15.609)	42.439 (20.462)	31.667 (17.658)
Range	2.000–299.000	1.000–282.000	2.000–168.000	1.000–299.000
Fuel				
Mean (SD)	20.429 (6.370)			20.429 (6.370)
Range	1.000–125.000			1.000–125.000

2.6. Technical Efficiency Empirical Model

The production frontier model for the three vessel groups was specified as the translog production. The SFA model and prediction of technical efficiencies for the fishing fleet were then performed using R version 4.2.2 with packages plm applied to organize a panel structure of the data and frontier to run the SFA model [35–38].

3. Results

3.1. The Status and Trend of Vessel Types on Lake Victoria

Motorized vessels exhibited a consistent increase in numbers, with the highest count of these recorded in 2020 in all three countries: around 17,000 in Uganda, 12,000 in Tanzania, and 5000 in Kenya (Figure 2). In contrast, paddle vessel usage in Uganda and Tanzania displayed parallel fluctuations from 2000 to 2016, followed by a decline of 53% and 7%, respectively, in 2020. Conversely, Kenya experienced a distinct trajectory, with a 19% decrease from its 2006 peak of 8324 vessels to 6749 in 2020. Sailed vessels steadily declined in use across all three countries during the same period.

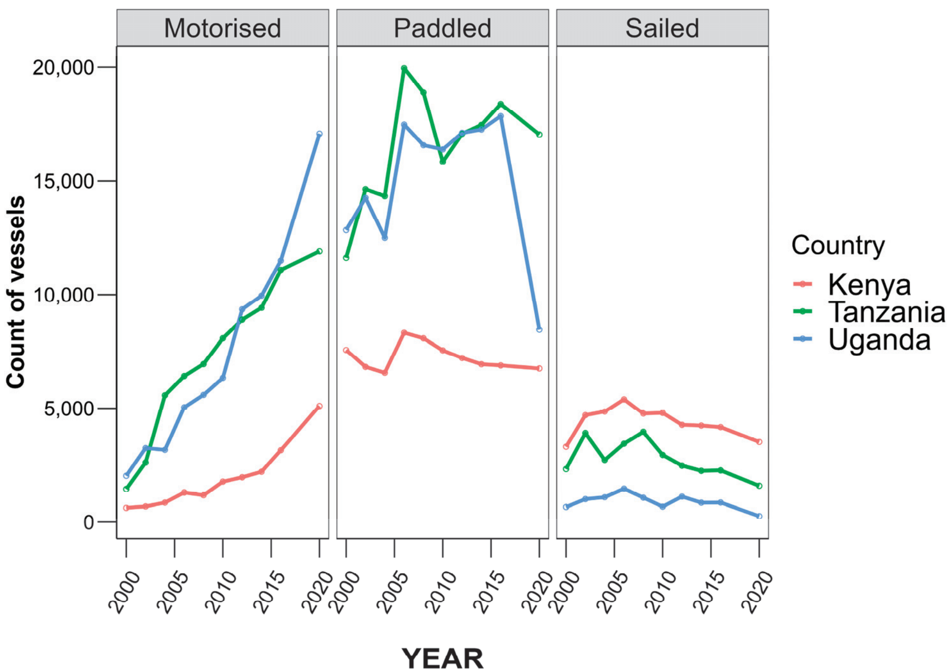

Figure 2. Status of vessel development by propulsion on Lake Victoria [7].

3.2. Technical Efficiency Estimation

Motorized fleets had the highest mean technical efficiency (0.60–0.66) compared to the paddled (0.29–0.49) and sailed vessel categories (0.24–0.46). Table 2 presents the estimates from the Translog stochastic production frontier analysis. The first-order parameters of vessel inputs (gear units, labor, and fuel for motorized vessels) and technical efficiency parameters (gamma and sigma squared) were all positive and significant across all vessel groups. These parameters represent output elasticities, with labor showing slightly higher elasticity than gear units and fuel. The gamma values were significant for all vessel groups (85% for motorized, 75% for paddled, and 70% for sailed vessels). The significant sigma squared σ^2 values further confirm the model's fit and the correctness of the composite error term's distributional assumption.

Table 2. Maximum likelihood estimates for the parameters of the stochastic frontier production function (SFPF).

SFA	Parameter	Motorized		Paddled		Sailed	
		Estimate	Standard Error	Estimate	Standard Error	Estimate	Standard Error
Intercept	β_0	0.217	0.022 (***)	0.685	0.032 (***)	0.967	0.043 (***)
InUnits	β_1	0.158	0.005 (***)	0.169	0.0058 (***)	0.102	0.009 (***)
Ifuel	β_2	0.089	0.030 (**)				
ILabor	β_3	0.317	0.017 (***)	0.294	0.014 (***)	0.216	0.017 (***)
I(0.5 * InUnits^2)	β_{11}	0.071	0.007 (***)	0.042	0.004 (***)	0.007	0.008
I(0.5 * Ifuel^2)	β_{22}	−0.036	0.044				
I(0.5 * ILabor^2)	β_{33}	0.338	0.038 (***)	−0.089	0.026 (***)	−0.096	0.033 (**)
I(InUnits * Ifuel)	β_{13}	−0.034	0.020				
I(InUnits * ILabor)	β_{12}	0.086	0.012 (***)	0.038	0.007 (***)	−0.010	0.009
I(Ifuel * ILabor)	β_{23}	0.054	0.049				
Country-specific inefficiency effect							
Z_(Intercept)	z_0	−4.218	1.353 (**)	1.401	0.080 (***)	1.679	0.051 (***)
Z_CountryTanzania	z_1	−0.626	0.175 (**)	−1.292	0.082 (***)	−1.114	0.038 (***)
Z_CountryUganda	z_2	0.813	0.218 (***)	−0.966	0.067 (***)	−0.585	0.038 (***)
Variance variables							
sigmaSq	σ^2	3.391	0.656 (***)	1.561	0.072 (***)	1.128	0.026 (***)
gamma	γ	0.853	0.027 (***)	0.748	0.008 (***)	0.696	0.016 (***)

Significance denoted: 0 '***'; 0.001 '**'; 0.01 '*'.

The technical inefficiency model revealed significant z_0 values across all vessel groups, indicating country-specific inefficiencies. The signs of the z_1 and z_2 variables determined whether a vessel group was inefficient (positive sign) or efficient (negative sign). For instance, in Uganda, an increase in the number of motorized vessels was associated with increased inefficiency, while in Tanzania, more motorized vessels were likely to increase technical efficiency with respect to the Kenyan motorized vessels. For other vessel groups, negative z-variable signs indicated a reduction in inefficiency as vessel numbers increased.

3.3. Technical Efficiency Distribution and Change

Technical efficiency score indicates that vessels are fully efficient at score 1 and inefficient tending to 0. From the TE estimation, vessels across all groups were inefficient as the maximum efficiency values were less than 0.90 (Table 3). Across countries, the estimated mean TE values for all vessel groups were highest in Tanzania, while Uganda had the lowest estimated mean TE for motorized vessels. The lowest mean TE values for paddled and sailed vessels were recorded in Kenya.

Table 3. Mean TE values per country and vessel propulsion.

Country	Kenya				Tanzania				Uganda			
Statistic	Mean	Max	Min	N	Mean	Max	Min	N	Mean	Max	Min	N
Paddled	0.290	0.800	0.037	1463	0.490	0.880	0.036	13,719	0.440	0.870	0.045	11,877
Sails	0.240	0.830	0.030	9087	0.460	0.890	0.020	8283	0.350	0.830	0.040	2259
Motorized	0.640	0.89	0.06	2372	0.660	0.890	0.050	12,406	0.600	0.870	0.040	15,260

The TE distribution shows that at least 63% of motorized vessels operated with efficiency levels above 0.6. A similar proportion of paddled vessels operated from >0.41, while sailed vessels of the same proportion operated at <0.40, indicating that the latter were utilizing less than half of their capacity to maximize catches (Figure 3).

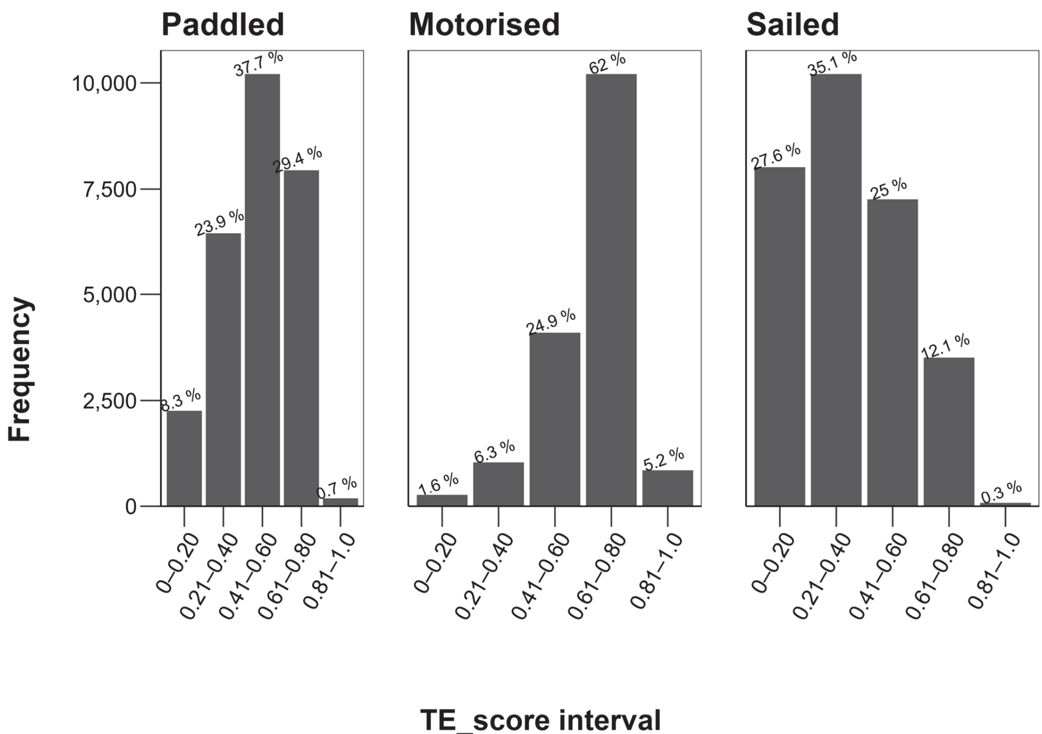

Figure 3. Distribution of TE scores for the three vessel groups, with scores categorized into groups.

Exploring the variations in technical efficiency (TE) throughout the study period (Figure 4) shows that both Uganda and Tanzania witnessed a noticeable reduction in technical efficiency (TE) across all vessel groups. The null hypothesis that there is no country-specific technical inefficiency was tested for each fleet segment. The hypothesis was always rejected, indicating that country-specific inefficiency differences exist for all vessel types. The most significant and consistent decline was observed in Ugandan motorized vessels, where the capacity to maximize catches for their given input and technology dropped by 22%, decreasing from 0.65 TE in 2005 to 0.50 in 2021. In Kenya, TE showed variations among different vessel types. Paddled vessels demonstrated an improvement in TE, increasing from 0.24 in 2011 to 0.38 in 2021, marking a substantial 50% enhancement in efficiency for this vessel category. Motorized vessels, on the other hand, exhibited a modest 2% increase in TE, while TE for sailed vessels declined by 12% between 2015 and 2021.

3.4. Labor Productivity

Labor productivity, defined as catch per hour fished, serves as a measure of fishers' productivity. From 2005 to 2015, all vessel groups experienced minor fluctuations in labor productivity. However, a significant increase in labor productivity was observed for motorized and paddled vessels in Kenya from 2015 to 2021 (Figure 5). In contrast, the same categories of vessels in Uganda and Tanzania showed minimal changes, with a slight downward trend from 2015 to 2020. Sailed vessels maintained a consistent level of labor productivity across all countries throughout the entire study period.

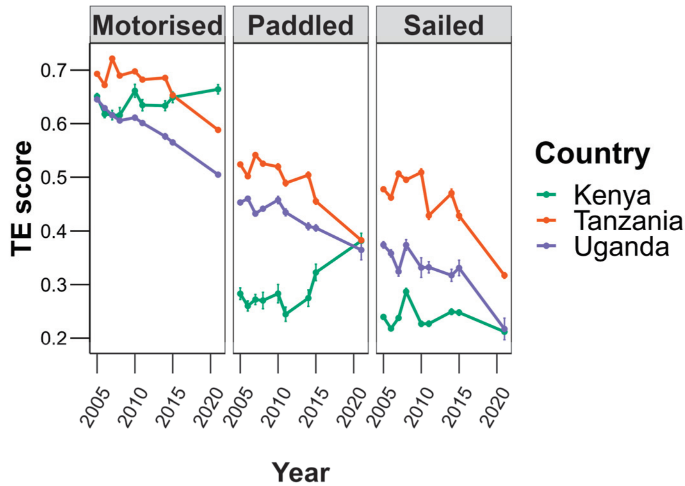

Figure 4. Technical efficiency (TE) estimates across vessel groups and countries over the study period.

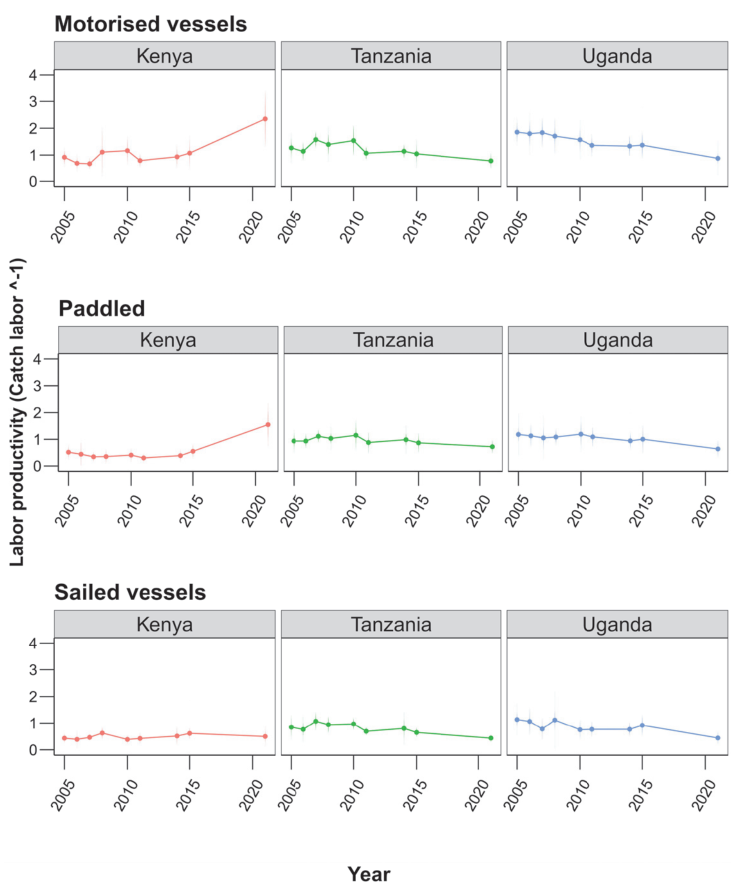

Figure 5. Comparison of labor productivity (LP) by vessel type and country (2005–2015).

4. Discussion

4.1. Vessel Group Fleet Changes and Technical Efficiency (TE)

The results show that efficiency and productivity vary between vessel groups, countries, and over time. This sheds light on the effects of technological development, policies, and natural and social conditions on economic outcomes in the fishing industry.

The finding that motorized fleets had the highest mean technical efficiency compared to the paddled and sailed vessel categories aligns with the research conducted by Kateregga and Sterner [14] in Lake Malawi [22], who highlighted the significance of vessel motorization in enhancing fish productivity. Similar effects have been observed by Branch et al. [39] regarding the motorized Fanti vessels in Liberia compared to their unmotorized counterparts [40]. The natural progression within the fishing industry is that fishers strive for more efficient operations, and this is evidenced by the general increase in the number of motorized vessels for all of Lake Victoria's riparian countries.

Motorized and paddled vessels form the two most important vessel types in the fishery. While the significant decline in paddled vessels in Uganda and Tanzania from 2016 to 2020 can be attributed to rigorous enforcement efforts to eradicate illegal fishing gear, the consistent decrease in these vessels in Kenya warrants further investigation to uncover the underlying factors.

On the other hand, sailed vessels have been declining in number. The low TE values in Kenya (<0.30) may explain the decline in sailed and paddle vessels. Motorized vessels across the three countries, with a TE of ≥ 0.60, could boost catches by 40% on average with current technology, while sailed vessels, the least efficient, have on average over 60% capacity for improvement. The motorized and paddled vessel groups were earlier described as commercial and artisanal, respectively, on Lake Victoria in Uganda [41,42]. Therefore, the shift to commercial fisheries indicates a within-sector improvement toward productivity growth [43–45], as fishers shift to motorized fishing vessels, the most technically efficient vessel type.

4.2. Comparative Analysis of Technical Efficiency and Labor Productivity across Countries and Vessel Groups

Country-specific comparisons show a difference in fleet development in Kenya versus Uganda and Tanzania. A shift towards more commercial vessel operations was observed across all countries, which is indicative of a boost in vessel productivity over time. In Tanzania and Uganda, the pattern for artisanal (paddled) vessels was similar, showing fluctuations for the first 15 years and a sharp decline between 2016 and 2020 due to fisheries enforcement. In contrast, Kenya has seen a consistent decrease in the use of paddled vessels since 2008, even though efficiency has been improving for this segment, a unique development for the fishery. This difference might be due to several factors, such as differences in fisheries management, economic development, and the different alternative values of labor in the three countries [39]. The three counties have had different fisheries policies in effect during the period. However, the most stringent policies have been found in Uganda and Tanzania, where technical efficiency and labor productivity have declined between 2015 and 2020. At the same time, improvements in technical efficiency and labor productivity were observed in Kenya. It is therefore difficult to attribute the development to fisheries management. Other forces could be at play. For example, countries where the population is large relative to capital and natural resources, the most productive sectors of the economy, are likely to have negligible to zero marginal productivity of labor and declining labor productivity, as is indicated in Uganda and Tanzania in this study [43,44]. Labor productivity results can highlight trends in labor markets such as increasing or decreasing employment and skills indicative of economic sustainability for the fishers; however, further analysis is needed on this issue.

The objective of the stringent fisheries enforcement in Uganda and Tanzania was to raise stock sizes of Nile perch by reducing illegal fishing activity, thereby improving fish exports [45]. The reported biomass estimates for the Nile perch before and after enforcement have followed a similar variable trend for all countries. Gear size and type had a small influence on fish stocks, as the Kenyan side maintained its biomass [8]. Successful policy implementation should lead to improved vessel efficiency, but the evidence for such effects regarding Uganda and Tanzania is weak [8,11,45–47]. This is in line with substantial literature that shows that policies that prioritize maximizing productivity may negatively impact the long-term sustainability of fish stocks, leading to depletion and even collapse of certain species [48–50]. The Ugandan and Tanzanian model of fisheries management illustrates the difficulty of regulating activities that people are compelled to undertake given their negative economic situation.

While the study demonstrates that the data used in this study can be effectively used to assess fishing fleet performance and inform fishery management in data-deficient contexts, it is important to recognize that the application of stochastic frontier production requires larger datasets to yield more robust results. As such, interpretations should be approached with caution, given the potential limitations in the data's scope of this study. Nevertheless, the findings still provide valuable insights into fleet efficiency and management strategies in resource-limited fisheries.

5. Conclusions

This study focuses on evaluating the technical efficiency of the Nile perch fishing fleet on Lake Victoria, categorizing vessels into three distinct groups based on their technology. Motorized vessels exhibited the highest efficiency (mean 0.60–0.66), showcasing their significant growth throughout the study. The declining trend observed in sailed vessels is reflected in their low technical efficiency across all countries, with specific variations observed for paddled vessels between Kenya and the other riparian countries.

The study acknowledges that vessel development mirrors the economic progress of the East African economies. The prevalence of paddled vessels in Uganda and Tanzania underscores their importance in artisanal fisheries, driven by a low opportunity cost of labor compared to Kenya. The improvement in technical efficiency and labor productivity in Kenyan vessels indirectly highlights gaps in fisheries management, questioning the effectiveness of enforcement, consideration of fish population status, and socio-economic conditions for alternative employment.

Overall, this study's primary contributions involve showing how sparse and deficient data can be utilized and interpreted in fisheries management, illustrating the application of technical efficiency in evaluating economic outcomes and fish stock health. It emphasizes the importance of incorporating CAS data into econometric models for resource assessment and policy evaluation, underlining the significance of monitoring fishery statistics. By analyzing transboundary fisheries data from Kenya, Uganda, and Tanzania, this study offers a unique perspective on factors impacting these fisheries, contributing to comparative studies on fishery performance in the African Great Lakes region.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/fishes9100414/s1, Table S1: Summary statistics of fuel model key variables; Table S2: Fuel model regression results for motorized vessels in Kenya. Remba and Kokach are landing sites; Table S3: Fuel model regression results for motorized vessels in Uganda; initial starting with L represents selected landing sites i.e., L_NK for landing site Nakatiba, V_SF is for Vessel type Ssese Flat and GG_Number is the number of gillnets; Table S4: Fuel model regression results for motorized vessels in Tanzania. G_GN for Gear_Gillnets, GG_number is Gillnets number.

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References

1. Nyamweya, C.S.; Nyaboke, H.M.; Aura, C.M.; Momanyi, K.N.; Mlaponi, E.; Odoli, C.O.; Njiru, J.M. Lake Victoria's bounty: A case for riparian countries' blue economic investment. *Front. Environ. Sci.* **2022**, *10*, 952654. [CrossRef]
2. LVFO. *Regional Catch Assessment Survey for Lake Victoria*; LVFO: Jinja, Uganda, 2022.
3. Njiru, J.; van der Knaap, M.; Kundu, R.; Nyamweya, C. *Lake Victoria Fisheries: Outlook and Management*; John Wiley and Sons Inc.: Hoboken, NJ, USA, 2018. [CrossRef]
4. Samy-Kamal, M.; Teixeira, C.M. Diagnosis and Management of Small-Scale and Data-Limited Fisheries. *Fishes* **2023**, *8*, 39. [CrossRef]
5. Taabu-Munyaho, A.; Marshall, B.E.; Tomasson, T.; Marteinsdottir, G. Nile perch and the transformation of Lake Victoria. *Afr. J. Aquat. Sci.* **2016**, *41*, 127–142. [CrossRef]
6. Taabu-Munyaho, A.; Kayanda, R.J.; Everson, I.; Grabowski, T.B.; Marteinsdóttir, G. Distribution and exploitation of Nile perch Lates niloticus in relation to stratification in Lake Victoria, East Africa. *J. Great Lakes Res.* **2013**, *39*, 466–475. [CrossRef]
7. LVFO. *Lake Victoria Frame Survey Report*; LVFO: Jinja, Uganda, 2020.
8. Nyamweya, C.S.; Natugonza, V.; Kashindye, B.B.; Mangeni-Sande, R.; Kagoya, E.; Mpomwenda, V.; Mziri, V.; Elison, M.; Mlaponi, E.; Ongore, C.; et al. Response of fish stocks in Lake Victoria to enforcement of the ban on illegal fishing: Are there lessons for management? *J. Great Lakes Res.* **2023**, *49*, 531–544. [CrossRef]
9. Kudhongania, A.W.; Chitamwebwa, D.B.R. Impact of environmental change, species introductions and ecological interactions on the fish stocks in Lake Victoria. In *The Impact of Species Changes in African Lakes*; Pitcher, T.J., Hart, P.J.B., Eds.; Chapman & Hall: London, UK, 1995; pp. 19–32.
10. Okeyo, D.O. Artisanal and commercial fishing gear and practices in the Lake Victoria basin drainage systems of Kenya: A photodiagrammatic verification. *Lakes Reserv.* **2014**, *9*, 192–205. [CrossRef]
11. Mpomwenda, V.; Tomasson, T.; Pétursson, J.G.; Taabu-Munyaho, A.; Nakiyende, H.; Kristófersson, D.M. Adaptation Strategies to a Changing Resource Base: Case of the Gillnet Nile Perch Fishery on Lake Victoria in Uganda. *Sustainability* **2022**, *14*, 2376. [CrossRef]
12. Jul-Larsen, E. *Management, Co-Management, or No Management?: Major Dilemmas in Southern African Freshwater Fisheries*; Food & Agriculture Organition: Rome, Italy, 2003; Volume 422.
13. Tweddle, D.; Cowx, I.G.; Peel, R.A.; Weyl, O.L.F. Challenges in fisheries management in the Zambezi, one of the great rivers of Africa. *Fish Manag. Ecol.* **2015**, *22*, 99–111. [CrossRef]
14. Kateregga, E.; Sterner, T. Production Functions for Nile Perch and Tilapia Fisheries: A case study of Uganda's Section of Lake Victoria. *Crown J. Agric.* **2012**, *1*, 1–12.
15. Lokina, R.B. Technical efficiency and the role of skipper skill in artisanal Lake Victoria fisheries. *Environ. Dev. Econ.* **2009**, *14*, 497–519. [CrossRef]
16. Mkuna, E.; Baiyegunhi, L.J.S. Impacts of Nile Perch (*Lates niloticus*) overfishing on the technical efficiency of fishers in the Tanzania portion of Lake Victoria. *Lakes Reserv.* **2021**, *26*, e12373. [CrossRef]
17. Bailey, C. Employment, labour productivity and income in small-scale fisheries of South and Southeast Asia. In Proceedings of the IPFC Symposium on Socio-Economic Issues in Coastal Fisheries Management, Bangkok, Thailand, 23–26 November 1993; pp. 24–45.
18. Ng'onga, M.; Kalaba, F.K.; Mwitwa, J. The contribution of fisheries-based households to the local economy (Capital and Labour) and national fish yield: A case of Lake Bangweulu fishery, Zambia. *Sci. Afr.* **2019**, *5*, e00120. [CrossRef]

19. Stanford, R.J.; Wiryawan, B.; Bengen, D.G.; Febriamansyah, R.; Haluan, J. Improving livelihoods in fishing communities of West Sumatra: More than just boats and machines. *Mar. Policy* **2014**, *45*, 16–25. [CrossRef]
20. Morrison, P.; Catherine, J.; Felthoven, R.G.; Torres, M.d.O. Productive performance in fisheries: Modelling, measurement, and management. *Aust. J. Agric. Resour. Econ.* **2010**, *54*, 343–360. [CrossRef]
21. Pascoe, S.; Mcwhinnie, S.; Schrobback, P.; Hoshino, E.; Curtotti, R. Using Productivity Analysis in Fisheries Management: A Guide for Managers and Policy Makers. 2019. Available online: www.frdc.com.au/project/2019-026 (accessed on 25 May 2024).
22. Simfukwe, K.; Limuwa, M.M.; Njaya, F. Are Chilimira Fishers of Engraulicypris sardella (Günther, 1868) in Lake Malawi Productive? The Case of Nkhotakota District. *Sustainability* **2022**, *14*, 16018. [CrossRef]
23. Channel, E.; Pascoe, S.; Coglan, L. The Contribution of Unmeasurable Inputs to Fisheries Production: An Analysis of Technical Efficiency of Fishing Vessels in the English Channel. *Am. J. Agric. Econ.* **2002**, *84*, 585–597.
24. Kirkley, J.E.; Squires, D. *Capacity and Capacity Utilization in Fishing Industries*; University of Carlifornia: San Diego, CA, USA, 1999.
25. Pascoe, S.; Mardle, S. Econometric measurement of technical efficiency. In *Efficiency Analysis in EU Fisheries: Stochastic Production Frontiers and Data Envelopment Analysis*; Pascoe, S., Mardle, S., Eds.; University of Portsmouth: Portsmouth, UK, 2003; pp. 25–42.
26. Crisci, A.; Siviero, L.; Ambra, L.D. Technical efficiency with several stochastic frontier analysis models using panel data. *Electron. J. Appl. Stat. Anal.* **2016**, *9*, 736–759. [CrossRef]
27. Aigner, D.; Lovell, C.A.K.; Schmidt, P. Formulation and estimation of stochastic frontier production function models. *J. Econom.* **1977**, *6*, 21–37. [CrossRef]
28. Huang, C.J.; Liu, J.-T. Estimation of a Non-Neutral Stochastic Frontier Production Function. 1994. *J. Product. Anal.* **1994**, *5*, 171–180. [CrossRef]
29. Battese, G.E.; Corra, G.S. Estimation of a production frontier model: With application to the pastoral zone of Eastern Australia. *Aust. J. Agric. Econ.* **1997**, *21*, 169–179. [CrossRef]
30. LVFO. *Standard Operating Procedures (SOPs) for Catch Assessment Surveys on Lake Victoria*; LVFO Standard Operating Procedures No. 3; LVFO: Jinja, Uganda, 2007.
31. LVFO. *The Standard Operating Procedures for Catch Assessment Surveys*; LVFO: Jinja, Uganda, 2021.
32. Verbeek, M. Pseudo-Panels and Repeated Cross-Sections. *SSRN Electron. J.* **2007**, *46*, 369–383. [CrossRef]
33. Pascoe, S.; Mardle, S. *Single Output Measures of Technical Efficiency in EU Fisherie*; University of Portsmouth: Portsmouth, UK, 2003.
34. Henningsen, A. *Introduction to Econometric Production Analysis with R*; Leanpub: Copenhagen, Denmark, 2014.
35. Croissant, Y.; Millo, G. Panel data Econometrics: The plm Package. *J. Stat. Softw.* **2008**, *27*, 1–43. [CrossRef]
36. Coelli, T.; Henningsen, A. Frontier: Stochastic Frontier Analysis. R Package Version 1.0. Available online: https://cran.r-project.org/web/packages/frontier/index.html (accessed on 15 October 2024).
37. James, G.M.; Witten, D.; Hastie, T.; Tibshirani, R. *An Introduction to Statistical Learning: With Applications in R*; Springer: New York, NY, USA, 2014.
38. Team, R.C. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2024; Available online: https://www.R-project.org/ (accessed on 10 October 2024).
39. Branch, T.A.; Hilborn, R.; Haynie, A.C.; Fay, G.; Flynn, L.; Griffiths, J.; Marshall, K.N.; Randall, J.K.; Scheuerell, J.M.; Ward, E.J.; et al. Fleet dynamics and fishermen behaviour: Lessons for fisheries managers. *Can. J. Fish. Aquat. Sci.* **2006**, *63*, 1647–1668. [CrossRef]
40. Jueseah, A.S.; Tómasson, T.; Knutsson, O.; Kristófersson, D.M. Technical efficiency analysis of coastal small-scale fisheries in Liberia. *Sustainability* **2021**, *13*, 7767. [CrossRef]
41. Mpomwenda, V.; Kristófersson, D.M.; Taabu-Munyaho, A.; Tómasson, T.; Pétursson, J.G. Fisheries management on Lake Victoria at a cross-roads: Assessing fishers' perceptions on future management options in Uganda. *Fish Manag. Ecol.* **2021**, *2*, 12526. [CrossRef]
42. Funge-Smith, S. *Review of the State of the World Fishery Resources: Inland Fisheries*; Food and Agriculture Organisation of the United Nations (FAO): Rome, Italy, 2018.
43. Sadik-Zada, E.R. Natural resources, technological progress, and economic modernization. *Rev. Dev. Econ.* **2021**, *25*, 381–404. [CrossRef]
44. Lewis, W.A. *Economic Development with Unlimited Supplies of Labour*; Manchester School of Economic and Social Studies: Manchester, UK, 1954.
45. Mpomwenda, V.; Tómasson, T.; Pétursson, J.G.; Kristófersson, D.M. From Co-Operation to Coercion in Fisheries Management: The Effects of Military Intervention on the Nile Perch Fishery on Lake Victoria in Uganda. *Fishes* **2023**, *8*, 563. [CrossRef]
46. Asche, F.; Smith, M.D. Viewpoint: Induced Innovation in Fisheries and Aquaculture. *Food Policy* **2018**, *76*, 1–7. [CrossRef]
47. Eigaard, O.R.; Marchal, P.; Gislason, H.; Rijnsdorp, A.D. *Technological Development and Fisheries Management*; Taylor and Francis Inc.: Abingdon, UK, 2014. [CrossRef]
48. Jacobs, F.J.; Naesje, T.F.; Ulvan, E.M.; Hay, C.J.; Khaebeb, F.H.; Weyl, O.L.F. Evidence for overfishing of tigerfish Hydrocynus vittatus (Castelnau 1861) in the Kavango River, Namibia. *Afr. J. Aquat. Sci.* **2022**, *47*, 207–215. [CrossRef]

49. Pomeroy, R.S. Managing overcapacity in small-scale fisheries in Southeast Asia. *Mar. Policy* **2012**, *36*, 520–527. [CrossRef]
50. Squires, D.; Vestergaard, N. Technical change and the commons. *Rev. Econ. Stat.* **2013**, *95*, 1769–1787. [CrossRef]

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Review

Exploring Policy of Small-Scale Coastal Fisheries in China: Evolution, Challenges and Prospects

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Abstract: China plays a significant role in the global fishing industry. The small-scale fisheries (SSFs) operating along its coast have made noteworthy and invaluable contributions in the areas of poverty alleviation, protein provision, social equity, and overall socioeconomic development. Coastal small-scale fishing management is a persistent challenge for all fishing nations, including China. In recent years, China has made significant strides in adopting scientific and refined approaches to fishery management in this sector. This paper provides an overview of the development of China's coastal fishery management practices, including changes in policies, methods, and modes since the establishment of the People's Republic of China (PRC) in 1949. To address these challenges, this research seeks to enhance the governance system of small-scale coastal fisheries by assessing values from three dimensions: society, economy, and ecology.

Keywords: small-scale fisheries; policy evolution; coastal fisheries; sustainable fisheries policy; data-limited fisheries; China

Key Contribution: Small-scale coastal fisheries management involves multiple economic, social, and ecological objectives. Since the 1950s, China has implemented a series of policies to improve coastal fisheries management. Focusing solely on a single value dimension and management approach cannot address the decline in coastal fishery resources and poses challenges to the livelihoods of traditional fishermen. To address these issues, the government should enhance awareness of sustainable coastal fisheries, expand policy focus beyond ecological value, strengthen understanding and research of SSFs, and implement comprehensive policy support measures that integrate economic, social, and ecological considerations. Enhancing the organizational level of coastal fishing communities, adopting mixed governance models, and employing diverse management approaches are crucial for the sustainable development of coastal fisheries. Continuous exploration and reform are guiding China's coastal fisheries management toward differentiation, refinement, and scientific approaches.

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1. Introduction

Over 90% of the 120 million fishers globally who rely on fishing for their livelihoods engage in SSFs [1]. These fisheries contribute to over half of the world's fish catch, with the majority (90–95%) being consumed locally [2]. However, the dominance of large industrial vessels in fisheries management has resulted in the neglect of policy considerations for SSFs [3]. Addressing the needs of SSFs requires global recognition and support. The Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries in the Context of

Food Security and poverty Eradication (abbreviated as SSF-Guidelines) [2] serve as a foundation and policy framework for promoting sustainable development, conservation of resources, and protection of rights and interests in coastal fisheries countries. Despite the guidelines, several challenges remain in their implementation, such as inadequate financial and technical resources from governments, the need for complementary policy support, and limited international cooperation. To ensure the sustainability of small-scale fishers' livelihoods and facilitate progress towards sustainable development, increased policy input and support are necessary in implementing the global action plan [4].

In China, the classification of coastal marine capture fisheries into SSFs/artisanal or commercial sectors is not straightforward. Factors such as engine size, length of vessels, distance from shore, water depth, and fishing gear used all contribute to the same type of catch [5]. Generally, China's coastal capture fisheries are considered small scale due to their concentration in inshore waters, where fishing vessels have distinct differences in size and power compared to medium and large-scale vessels. However, the 2019 revision of Provisions on the Administration of Fishing Licenses (hereafter referred to as PAFL, 2019) does not provide a clear definition of SSFs. It only distinguishes small-scale marine fishing vessels without offering a comprehensive understanding of the concept [6]. The management of small-scale fishing vessels and their operations currently lack cohesion and defined objectives. Furthermore, the absence of corresponding content in legal and policy documents at provincial and county levels in coastal regions poses additional challenges in defining SSFs.

In light of these circumstances, the 14th Five-Year Plan for Fisheries Development was introduced in January 2022 [7]. This plan includes adjustments in the allocation of fishery resources, such as limiting the number and area of fishery licenses, implementing fishery quotas, and enforcing seasonal fishery closures. It also supports the transformation and modernization of small-scale fishing vessels while strengthening supervision and law enforcement. The promotion of cooperation between the central and local governments, fishery associations, and fishermen's organizations indicates a shift towards a more differentiated, refined, and scientific approach to fishery management by the central government.

As the 14th Five-Year Plan for Fisheries Development nears its conclusion, we have conducted an analysis of China's coastal fisheries policy changes over the past 75 years. This analysis aims to understand the motivations behind the policies, assess their effectiveness in implementation, and identify existing issues. The findings will facilitate the formulation of more precise and scientifically grounded policy recommendations for coastal fisheries management in the upcoming 15th Five-Year Plan.

2. Material and Methods

2.1. Policy Text Analysis

We use the policy text analysis method and have a comprehensive and overall grasp of the policy text. First of all, we manually collected the laws, administrative regulations, departmental rules, normative documents, and fishery development plans related to the coastal fishing industry promulgated by the government of China from 1949 to 2024 through the keywords "Coastal fishing industry" or "Small fishing vessels". We paid attention to the objectives, scope, types, and policy tools involved in the policies. Combined with the collected policies, we can infer the relevant characteristics of policy evolution [8].

2.2. Documentary Analysis

We conducted an analysis of 27 papers using the keywords "China coastal fishing fishery" and "fishery policy" within the Web of Science database and made clear the core policies and characteristics of the coastal fishing industry in China at different stages. In China, SSFs are not managed indiscriminately compared with medium and large-scale fisheries. Therefore, the policy characteristics of SSFs are similar to these, which can be roughly divided into the germination period (1949–1977), the development period (1978–2000), and the control period (2000–present).

3. Results

3.1. The Evolution and Characteristics of the Coastal SSFs Policy in China

3.1.1. Embryonic Period of SSFs Management (1949–1977)

Following the establishment of the People's Republic of China, rural areas in China remained largely reliant on traditional small-scale peasant production, characterized by limited productive capacity and low living standards. Amid the national emphasis on increasing grain output, the fishery sector has gained attention as a crucial factor in achieving this goal. Aquatic products serve as a significant protein source for humans. To mobilize collective efforts and maximize production, scattered and relatively unproductive SSF producers were integrated through the establishment of fishery cooperatives. These cooperatives primarily managed low-power wooden fishing boats. By 1958, with the establishment of the Fishery People's Commune, individual production and management by small-scale fishermen transitioned to collective management [9,10]. Fishery organizations, mainly in the form of fishery communities, were implemented, introducing a flexible system in terms of organizational scale, ownership, distribution, operation, and management, which facilitated increased fishery production. Additionally, designated coastal fishing ban zones were established to protect fishery resources in coastal waters.

3.1.2. Development Period of SSFs Management (1978–2000)

Since the Third Plenary Session of the 11th CPC Central Committee in 1978 opened the transition from China's planned economy to a market economy, great changes have taken place in fishery rights and fishery management systems. In particular, in the fishing areas of Guangdong and Fujian, where China's fishery is relatively developed, the coastal areas of Guangdong and Fujian have first begun to explore the reform of the fishing fishery management system. Haifeng County and Yangjiang County in Guangdong, Lianjiang County and Dongshan County in Fujian Province have successively started the reform of the fishing fishery management system and tried to offer a variety of new fishery management models such as discounting the price and accounting for cooperative fishing vessels [11], breaking the original management system of basic accounting with "team" as the production unit, and transferring the main means of production such as fishing boats and fishing nets from collectives to new economies such as cooperative fishing vessels through paid transfer. Fishermen have equal rights, joint possession, and domination. By 1983, the shareholding ratio of cooperative fishing vessels had risen from 15% to 70%, which meant that the fishing rights of the original production teams were generally transferred [11]. In the major fishing areas, we have comprehensively carried out the reform of the shareholding system for fishing vessels, discounted the prices of the fishing boats of the former brigades and gave them to the fishermen for joint ownership, turned large collectives into small groups, and implemented a system of accounting by boats and decentralized management. This is the change in the small-scale fishery management system from collective to individual, from centralization to decentralization in China. Because of China's vast land, long coast, and huge fishing vessels that operate less than 12 m along the coast, this is one of the factors that make it difficult to manage SSFs in China from collective to individual and from centralized to decentralized.

The decentralized management system has boosted fishermen's enthusiasm for production, but it has also led to an increase in the number of fishing vessels and dispersion of fishing activities, exacerbating the depletion of coastal fishery resources. At this stage, the Chinese government recognized the issue of declining fishery resources and implemented specific legislative measures to address it. These measures included the protection of aquatic resources, the implementation of a fishery license system, the establishment of a fishing vessel management system, and the introduction of a series of management and conservation systems for fishery resources.

3.1.3. Management and Control of SSFs (2000-Present)

In 1987, the "double control"—an input control method aimed at reducing or limiting the total fishing capacity of a fishing fleet by controlling the overall vessel number and horsepower of the fishing fleet system for marine fishing vessels, which regulates the number and power of fishing vessels—was introduced with the Opinions on the Control Indicators for Coastal Fishing Motorized Fishing Vessels. However, due to weak implementation, the total number and power of fishing vessels in China show a growing trend from 1992 to 1999, reaching a peak in 2000. During the Ninth Five-Year Plan period, the growth rate of fishing vessel power in the country declined sharply from 43% to 37%, and the number of fishing vessels experienced negative growth [12]. This decline was mainly attributed to a reduction in low-power fishing vessels, indicating that the fishing vessel and power control systems have helped to curb the excessive growth of fishing vessels.

In the 1990s, the excessive fishing activities in China's coastal areas led to a significant decline in the production of economically important fish species, with some resources approaching extinction. To address this issue and prevent overfishing, the Ministry of Agriculture introduced the concept of "Zero growth of marine fishing output" (hereafter referred to as Zero growth). A year later, the target of achieving "Negative growth of marine fishing output" (hereafter referred to as Negative growth) in marine capture production was set as a means to reduce the intensity of fishing and strengthen the protection and restoration of coastal fishery resources [13].

During this period, there was a shift in the fishery management system from the contract responsibility system to the shareholding system. This transition allowed for self-management and self-accountability in terms of profits and losses, which has remained in place until now [14].

In 2003, the Fisheries Bureau of the Ministry of Agriculture introduced the Opinions on the implementation of Marine Motorized Fishing Vessels Control System from 2003 to 2010, marking the start of the "total control system" phase during the Ninth Five-Year Plan, focusing on reducing the number of fishing vessels [15]. In 2017, the Ministry of Agriculture issued a notice to further strengthen the control of domestic fishing vessels and implement comprehensive management of marine fishery resources, specifying the number and power of fishing vessels in China's coastal provinces from 2015 to 2020 [16]. While the "Zero growth of marine fishing output" and "Negative growth of marine fishing" policies differ in timing and background from the previous double control and double reduction policy system, the objective remains the same: to reduce the intensity of small-scale fishing vessels in coastal areas. In the 14th Five-Year Plan for Fisheries Development released in January 2022, the management objectives for the coastal fishing industry have shifted from "strict control" (the management focus has changed from simply strictly controlling the total amount of fishing to optimizing the fishing structure, including optimizing the number and scale of fishing vessels, promoting the upgrading of fishing methods and technologies, and protecting fishery resources and ecological environment) to "optimization of fishing structure", indicating a move towards differentiated, refined, and scientifically managed fishery management by the central government [7].

3.2. Common Characteristics of Policies at All Stages

Through the analysis above, we find that small-scale coastal fisheries, although adjusted according to different social, economic, and policy conditions, share the following common characteristics across different policy stages.

3.2.1. Focus on Ecological Value

Fisheries policies are formulated to tackle intricate social challenges. If the value orientation of these policies is oversimplified, it may overlook the diverse requirements of society across various developmental stages and societal groups. Consequently, such policies may lack adaptability to intricate realities and overly concentrate on a sole objective, often disregarding the significance of long-term sustainable development. Consequently, these policies can prove ineffective when confronted with intricate SSFs.

From the perspective of the different stages of China's small-scale fishery policy, namely the embryonic stage, the development period, and the control period, these stages can be classified based on the policy's value dimension. The value dimension specifically encompasses the economic, social, and ecological aspects of China's fishing industry policy [17]. The economic value primarily centers on the provision of aquatic products, enhancing the market value of fish and other cash crops related to aquaculture, and driving economic growth in fishing-related industries. The social value emphasizes factors such as maintaining stability in coastal areas, improving the living conditions of fishermen, and promoting employment opportunities for fishermen. The ecological value focuses on the conservation of marine living resources and the restoration of marine ecosystems [17]. However, it is noteworthy that the policies pertaining to SSFs in China predominantly prioritize ecological value, with limited emphasis on social and economic value, as indicated in Table 1.

Table 1. Policies, tools, and value dimensions of SSFs introduced at different stages in China. The references for the policies are listed in Supplementary Table S1.

Period	Content Documents	Policy Objectives	Policy Tools	Value Dimension
Phase 1: Embryonic period (1949–1977).	Common Program of the Chinese People's Political Consultative Conference (1949) (The First Plenary Session of the Chinese People's Political Consultative Conference)	Preserving coastal fishing grounds and promoting aquaculture development.	Command-control	Ecological value
	Directives on fishers' work (1952) (Administrative Department for Fisheries, ADF)	Fishery production serves as a primary source of income for China's national economy and holds a prominent position among the country's top five agricultural industries.		Economic value
	Resolution on agricultural production cooperatives (1953) (Communist Party of China, CPC)	The development path of farmers from mutual aid groups and primary cooperatives to advanced cooperatives		Social value
	Order on the prohibition zones for motor trawl fishing in the Bohai Sea, the Yellow Sea, and the East China Sea (1955) (The State Council, People's Republic of China, PRC State Council)	Ensuring the conservation of China's coastal aquatic resources and minimizing conflicts between trawling and small-scale fishing activities.		Ecological value
Phase 2: Development period (1978–2000).	Regulations on the protection of aquatic resource reproduction (1979) (PRC State Council)	Conservation and preservation of aquatic resources through reproduction and protection measures.	Command-control	Ecological value
	Interim provisions on certain issues concerning fisheries Licensing (1979) (National ADF)	Implementing a fishery license management system to promote the rational and sustainable utilization of fishery resources and mitigate the risks of overfishing and resource depletion.		
	Notice on the summer fishing moratorium for collective trawlers and joint inspection of the proportion of juvenile fish on state-owned fishing vessels (1980) (National ADF)	Ensuring the conservation of fishery resources and preventing overfishing.		
	Decision of the State Council on the establishment of juvenile fish reserves (1981) (PRC State Council)	To safeguard the reproductive success and growth of *L. crocea* and *T. lepturus* juvenile		
	Report on several issues concerning the development of marine fisheries (1983) (National ADF)	Putting emphasis on the conservation of Coastal resources and regulating fishing intensity.		
	Directives on relaxing the policy and accelerating the development of the fishery and aquaculture industry (1985) (No. 5 Central Document) (CPC, PRC State Council)	Standardizing and modernizing the fishing industry will enhance the economic benefits and sustainable development capacity of fisheries.		Economic and ecological value
	Fisheries Law of PRC (1986, revised and reacted in 2000, 2004, 2009, 2013. The National People's Congress Standing Committee, NPCSC)			

Table 1. *Cont.*

Period / Content	Documents	Policy Objectives	Policy Tools	Value Dimension
	Provisions on the arrangement and management of fishing seasons in the main fishing grounds of the East China Sea, Yellow Sea, and Bohai Sea and Opinions on control indicators for nearshore and coastal motorized fishing vessels (1987) (National ADF)	To protect and rationally utilize key economic fish and shrimp resources, maintain fishing ground productivity, and regulate fishing intensity.		
	Opinions on controlling the growth indicators of marine fishing intensity during the 8th FYP period (1992) (National ADF)	Control the blind growth of nearshore fishing vessels and protect and rationally utilize fishery resources.		
	Opinions on the implementation of controlling indicators of marine fishing intensity during the 9th FYP period (1997) (National ADF)	Control the uncontrolled growth of coastal fishing vessels and ensure the protection and rational utilization of fishery resources.		Ecological value
	Opinions on further accelerating fisheries development (1997) (National ADF)	Enhance the protection and sustainable utilization of coastal fishery resources.		
	"Zero-growth" policy for national annual marine fishing yield (1999) (National ADF)	The shift from quantity expansion to quality and efficiency improvement has effectively enhanced the protection of fishery resources and the ecological environment.		
	"Negative growth" policy for annual fishing yield (2000) (National ADF)	Maintain the current level of marine fishing output in China.		
Stage 3: Restriction and compression period (2001–present).	Opinions on the implementation of Marine Motorized Fishing Vessels control system in 2003–2010 (2003) (National ADF)	Regulate the number and capacity of fishing vessels to ensure that fishing intensity in the area aligns with the sustainable catch limits of fishery resources.	Command-control	Ecological value
	Interim Provisions on the Use and Management of Special Funds for the Restructuring of Marine Fishermen (2004) (The Ministry of Finance of the People's Republic of China, MOF; MARA)	Encourage fishermen to change their professions and promote the optimization and upgrading of fishery industrial structures.	Financially incentive	Economic value
	Notice on the investigation of fishing gears and methods in the national fishing industry (2009) (The Ministry of Agriculture and Rural Affairs of the People's Republic of China, MARA)	Strengthen and standardize the management of fishing gear and fishing methods, control fishing intensity, and avoid overfishing.		
	Regulations of the People's Republic of China on the registration of fishing vessels (revised and reacted in 2013, MARA)	Strengthen the supervision and management of fishing vessels.	Command-control	Ecological value
	Notice on soliciting opinions on the improvement in the minimum mesh size system and the fishing gear access system for marine fishing 2013, MARA)	Protecting fishery resources and strengthening management of fishing gear.		

Table 1. *Cont.*

Period	Content	Documents	Policy Objectives	Policy Tools	Value Dimension
		Notice on further strengthening domestic fishing vessel management and implementing the system for managing total marine fisheries resources (2017, MARA)	Regulate the fishing capacity of vessels and catch, enhance the scientific and refined utilization and management of marine fishery resources, and achieve standardized and organized utilization of marine fishery resources.		
		Regulations on the administration of fishing permits (2019, MARA)	Classification, classification, and zoning control of fishing vessels		
		The 14th FYP for national fisheries development (2021, National ADF)	Enhance the organizational level of fishery production, implement scientific management of fishing vessels, and strengthen coastal law enforcement capabilities.		
		Notice on implementing fishery development support policies to promote high-quality development of fisheries (2021, MARA)	Optimize the fishery industry structure, reduce coastal fishing intensity, and protect marine resources.		

3.2.2. Mainly Based on Input Control Means

Command control is one of the commonly used policy tools [18]. In China, input control measures often need command-control tools to implement, which are commonly employed in the management of small-scale fisheries. This tool entails regulating the exploitation and utilization of fishery resources by controlling factors such as the number of individuals, fishing vessels, fishing gear, fishing grounds, fishing seasons, and fishery rights, employing government-led command and control mechanisms as a form of management [19,20]. In China, input policy tools (refer to Table 2) are characterized by the government's ability to enforce specific measures through administrative authority, particularly in public goods with pronounced negative externalities, such as SSFs, where it plays a crucial role in safeguarding coastal fishery resources and ensuring the safety of fishery production. However, in China, the command-control policy tools exhibit deficiencies in their policy design characteristics: (1) they overlook variations in the process of policy implementation. Command and control approaches often impose uniform standards for different types of fisheries in diverse regions with varying natural characteristics, scales, operating methods, and levels of development, thus underestimating and disregarding the socioeconomic impact of SSFs; (2) they fail to address information asymmetry. The governance challenges of SSFs stem from the difficulty in obtaining data, leading to the government's inability to acquire comprehensive, accurate, and complete information, resulting in decision making that is often flawed, thereby posing significant challenges [21,22].

Table 2. Management measures are commonly used in the management of SSFs in China.

Input Control	Purpose/Role
Management of fishing permits	Licenses serve as the only means of controlling the number of people involved in fishing.
Fishing efforts are controlled	There is a direct limitation of effort input, e.g., fishing time, traps, or trawl settings.
Closed to fishing, Closed fishing zone	conserving known populations at a specific place or time; it is usually its spawning or juvenile stage. It can also be used to control total fishing efforts, eliminating fishing from a specific group of areas or time of year.
Fishing gear restrictions	These are usually designed to control the size or type of fish being caught, for example, by adjusting the size of the mesh used in the net or trap.

3.2.3. Mainly Rely on the Management System of the Central Government

The existing SSFs management system, from the central to the local level, primarily focuses on controlling inputs based on quantitative indicators. However, there are several issues with the legal system and normative policies at all levels of government. Firstly, there is an unclear definition of SSFs and fishermen, which creates ambiguity and a lack of a solid management foundation. Secondly, local governments are not responsive to higher-level government policies. Local governments have been slow in implementing and adhering to central government policies. For instance, despite the new version of PAFL, coastal governments at all levels have not issued legal and policy documents related to the management of SSFs, nor have they made the necessary revisions in accordance with the new regulations on fishing permits.

In the current small-scale fishery management system, there is a lack of clear policy documents that provide a precise definition of SSFs. Differentiated policies are necessary to distinguish SSFs from other types of fisheries based on factors such as water type, dependency level, and historical evolution. Policy tools should be adjusted according to the specific characteristics of SSFs.

Regarding institutional arrangements, current policies have limited provisions addressing SSFs. Although PAFL has introduced several elements of SSFs management, there are no specific implementation rules at the local level. Generally, the physical characteristics, management models, and policy provisions for PAFL are similar. However, regional differences arise due to variations in the dependence of local fishermen on SSFs, economic income, aging levels, and government support [5].

Furthermore, the enforcement and regulatory capacity of local governments significantly impact the development of SSFs. In many areas, the lack of effective regulatory mechanisms and inadequate human resources allows unrestricted fishing, resulting in resource overexploitation and environmental degradation. The trade-off between economic development and environmental protection in some local governments may also affect the sustainable growth of SSFs.

The quality of fishermen and their understanding of policies also play a significant role in policy implementation. Many fishermen lack sufficient awareness of new policies, leading to deviations in their practices. Additionally, information asymmetry prevents some fishermen from accessing timely policy information, causing them to miss out on government support.

To address these challenges, it is crucial to establish a more scientific and adaptable local management system. This includes the development of implementation rules tailored to local contexts, increased awareness and training for SSFs, and improved policy understanding and participation among fishermen. Local governments should also enhance financial support, provide necessary technical guidance, and ensure resource availability to facilitate the sustainable development of SSFs.

While central policies provide a framework and guidance for small-scale fishery development, the active participation of local governments and fishermen is indispensable. Only through collaborative efforts can we promote the healthy and orderly growth of SSFs.

3.2.4. Command-Control Model Is Dominant

In some cases, command-control management models have certain advantages, such as improving management efficiency and the execution of decisions, ensuring the uniform implementation of policies and measures, and better centralizing and allocating limited resources. However, there are also some drawbacks to this management model, including relative rigidity, information asymmetry, lack of participation and feedback mechanism, and the social governance system being overly dependent on the government's command and control means and lacking autonomy and innovation. In addition, the command-control management model leads to a lack of financial support. Some local policies on small-scale fisheries, such as Shengsi County, have introduced more specific measures, but they cannot be implemented due to a lack of funds [5]. Due to China's command-control management model, the government's budget determines the government's priorities. Therefore, local governance issues do not attract attention, and neither do policy and financial support.

For example, according to the latest PAFL [6], the authority to approve the indicators of small fishing vessels and net tools is delegated to the provincial-level fishery authorities, and the provincial-level departments are decomposed and passed on layer by layer according to the resource status of the province and the goals and policy orientations set by the higher-level government. However, at present, local governments mainly focus on reducing fishing intensity, restoring coastal resources, promoting the conversion of fishing industries, and improving the capacity building of grassroots fishing vessels in accordance with the policy directives of the central government.

Stuck in the command-control management model for a long time, China has repeatedly demonstrated the ability to "concentrate on doing big things", which reflects the long-term dominance of command and control means in China's action logic, the social governance system has formed an over-dependence on the government's management model, and the social system is scattered and lacks experience and social trust.

4. Challenges and Constraints of SSFs Policy in China

Coastal fisheries have significantly contributed to China's economic development and the well-being of its people in recent decades. They have generated numerous employment opportunities for fishermen and related workers, thereby boosting income growth in coastal communities. In parallel, the Chinese government has prioritized marine ecological protection in coastal fisheries by implementing measures such as enhanced supervision, reduction in illegal fishing, prohibition of unauthorized fishing activities, and preservation of marine ecosystem diversity and stability.

Nonetheless, certain deficiencies in the governance of public affairs, particularly in small-scale fisheries, persist in China. These deficiencies include a lack of long-term and systematic policy support, limited management methods, and insufficient stakeholder participation. Consequently, small-scale fisheries often face challenges on the periphery of fishery management.

The PAFL introduced unprecedented regulations to oversee small fishing boats. These measures include provincial determination of the number of small fishing boats and restrictions on engine power, requirements for approval of fishing grounds in Class A waters (according to Article 23 of PAFL 2019, the marine fishing zone is divided into four categories: A, B, C, D. Class A includes the sea area of the Yellow Sea, Bohai Sea, East China Sea, South China Sea, and other sea areas on the land side of the outer boundary of closed fishing zone for bottom trawling by motorboat; Class B are Chinese and surrounding countries jointly managed fishing areas, which including Nansha sea area, Huangyan Island sea area and other specific fishery resources fishing grounds and aquatic germplasm conservation zone; Class C including the Bohai Sea, Yellow Sea, East China Sea, South China Sea and other sea areas under China's jurisdiction except Class A and Class B fishing zones. Class D fishing area is high seas), and limitations on each family obtaining fishing licenses for more than two small fishing boats. However, the government's understanding of the supporting role of SSFs in China's fisheries economics and industry remains limited.

4.1. Lack of a Sustainable Policy Framework

In the context of sustainable economic development, it is crucial to develop policies that encourage fisheries to adopt sustainable fishing techniques and management practices. This will ensure stable catches and long-term growth in the economic benefits of fisheries. Governments can provide financial support to promote the development and application of innovative technologies that reduce costs and improve the efficiency of fisheries operations. Additionally, policies should encourage the development of the fishery industry chain, provide employment opportunities, and promote economic growth and social prosperity. Developed countries have implemented various policies to support the economic growth of sustainable fisheries. For instance, the Government of Norway has introduced a system of catch quotas and fisheries permits to limit catches and ensure the sustainable use of fishery resources. They also encourage fishermen to adopt environmentally friendly fishing techniques, such as selective fishing and net selection devices, to reduce the catch of non-target species [23]. These measures not only improve fishing efficiency but also increase the economic benefits of fisheries.

In terms of sustainable social development, policies should focus on improving the living conditions of fishers and promoting the sustainable development of communities. Governments can provide training and skills upgrading programs to help fishers adapt to new fisheries management requirements. Social protection and welfare measures should also be implemented to safeguard the basic rights and welfare of fishers. Furthermore, policies should encourage the establishment of fishers' cooperatives and mechanisms for participation in fisheries decision making, enhancing community cohesion and autonomy [24–26]. Developed countries have implemented measures to improve the living conditions of fishermen and foster sustainable community development. For example, the Government of Iceland has implemented a training program for fishermen, providing them with the necessary skills and knowledge to adapt to new fisheries management requirements. Iceland has also introduced a cooperative fisheries management system that encourages the establishment of fishermen's cooperatives, fostering community cohesion and autonomy through consultation and participation in decision making [27].

In terms of ecologically sustainable development, policies should be implemented to protect fishery resources and maintain the health of marine ecosystems. Governments should formulate fishery management regulations, restrict catches, prohibit destructive fishing practices, and establish protected areas and moratoriums to restore and protect important fishery resources [28,29]. Strengthening monitoring and research efforts is also essential to understand the impacts of fisheries on ecosystems and adjust fisheries management policies based on scientific research. Developed countries have adopted a range of conservation measures to protect fishery resources and the health of marine ecosystems. For instance, the Government of Canada has established closed fishing areas and protected areas to restrict fishing activities and protect important fishery resources and ecosystems. Additionally, the Norwegian government has enhanced the monitoring of fishery activities, conducted scientific research to understand the impact of fisheries on ecosystems, and adjusted fisheries management policies accordingly to ensure the sustainable use of fishery resources [23].

4.2. Lack of Differentiated Policy Supply

We reviewed over 150 laws, administrative regulations, departmental rules, and normative documents related to fisheries in China, with a focus on the contents pertinent to SSFs (see Table 1). From a textual perspective, it is observed that most of these documents primarily center on the management of small fishing vessels, emphasizing the reduction in the number of fishing vessels and fishing intensity. Furthermore, the provisions related to the management of small-scale fisheries within China's relevant laws and policy documents are found to be quite limited. Access conditions for small-scale fishers, gear, and grounds have not been clearly defined. For instance, certain coastal provinces have not made corresponding adjustments to the content of fishing areas in marine Category A fishing areas for domestic marine small-scale fishing vessels, as outlined in the PAFL. Therefore, the operating areas of medium- and large-sized vessels and small-scale vessels continue to overlap. Additionally, traditional practices and species of small-scale fishermen have not been classified and managed. In addition to boats, manual types of work such as shore fishing, net insertion, shoveling, picking, and digging are also not adequately addressed.

4.3. Limited Data Are Difficult to Support Scientific Decision Making

For instance, in China's "People's Republic of China (PRC) Fisheries Law" and PAFL, it is stipulated that large-scale fisheries are required to fill in fishing logs, whereas small fishing vessels and manual fishing methods are not subjected to the same requirement. Furthermore, there are no other management requirements for information and data related to SSFs, neither at the national nor local level. This lack of data and dynamic information on the production process of SSF makes it difficult to achieve science-based management. Under the current Fisheries Law, the issuance of permits is supposed to be based on resource capacity indicators such as total allowable catches. However, permits are currently issued based on the number of applications by fishermen rather than the capacity of fishery resources. As a result, there are no substantial restrictions on the conduct of SSFs, making enforcement more challenging.

4.4. Insufficient Governance Capacity and Lack of Technical Assistance

The PAFL authority to approve indicators for small fishing vessels and net tools was delegated to provincial fishery authorities. This demonstrates that China's government has been transforming its functions in the context of decentralization, regulation, and service, aiming to reduce excessive intervention by the central government.

The management of SSFs primarily falls under the jurisdiction of towns and villages below the county level. However, specific fishers' organizations dedicated to SSFs have not yet been established. China's small-scale fishery sector continues to operate in a decentralized manner. The requirement for a fisherman's organization certificate to apply for a fishing license is currently replaced by the village committee or neighborhood committee to which the fisherman belongs. However, these organizations are not specifically fishers' organizations but general grassroots self-government organizations, and their members include all residents of villages and towns. Currently, there are no grassroots management organizations specifically dedicated to small-scale fisheries. Even if they exist, they are often managed in conjunction with small, medium, and large-scale fisheries. In some cases, SSFs are excluded by fisheries organizations to avoid increasing management costs. As a result, small-scale fishers have limited participation and influence within these organizations. When faced with management decisions made by the village committee, few opinions or suggestions are put forward by small-scale fishers, and even if they are, they are often ignored in line with the principle of majority rule.

The capacity of relevant central government agencies, local authorities, and small-scale fishers themselves is crucial for the effective implementation of policies. Training programs, technical assistance, and knowledge exchange platforms can play a significant role in building the necessary skills, knowledge, and capacity to address challenges in the implementation of governance policies.

4.5. Inadequate Law Enforcement Capacity

Limited enforcement capacity presents a significant challenge to the implementation of China's marine fisheries policies. The vast coastal areas and large numbers of small-scale fishermen make it difficult for government departments to effectively monitor and enforce regulations to ensure that fishing activities align with sustainable practices. Surveys conducted in Zhejiang Province in the East China Sea have revealed that local policies often fail due to challenges in implementation [5]. The decentralized and small-scale nature of fishing activities makes it challenging for regulatory authorities to cover all fishing vessels, resulting in ineffective monitoring and enforcement of sustainable fishing practices. Consequently, some fishing activities may not meet the requirements of sustainable fisheries, thereby impacting the marine ecological environment.

Another obstacle faced by law enforcement officials is the lack of a precise definition of SSFs and the absence of a uniform definition of small-scale fishing vessels. The inconsistent criteria for judging small-scale fisheries, as evidenced by the changing caliber of small-scale fishing vessel statistics in the Fishery Statistics Yearbook over the years, has led to a lack of standardization. The classification of small fishing vessels according to length, as introduced in the PAFL, has further contributed to a lack of uniformity in the description of small-scale fisheries among fishermen, law enforcers, and other stakeholders. This lack of clarity creates challenges in law enforcement due to the absence of a standardized basis for judgment.

5. Prospects for the Governance of SSFs in China

5.1. Establish a Policy Value System for SSFs

In order to develop effective fisheries policies, it is essential to consider the economic, social, and ecological values associated with fisheries. Economic value should be regulated and scientifically utilized through market mechanisms to ensure the sustainable use of fishery resources and establish effective supervision. Social value should be achieved by understanding and meeting the needs and expectations of fishers and coastal communities while also ensuring fair and equitable distribution of resources. Ecological value emphasizes the maintenance and protection of marine ecosystems' integrity and functioning (refer to Figure 1).

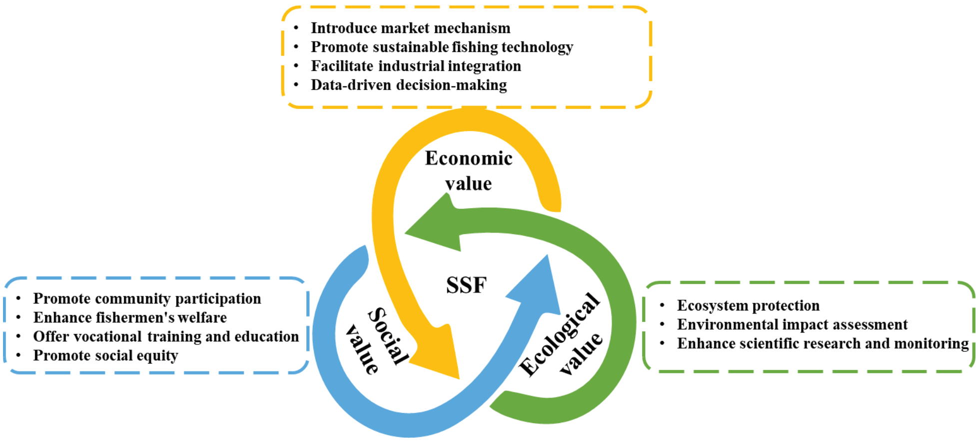

Figure 1. Value system of SSFs coastal sustainable fisheries policy.

To overcome the existing biases and imbalances in current approaches to fishery resource assessment and achieve social, economic, and institutional integration, several actions are necessary. Firstly, the assessment and management process should adopt a holistic approach that considers social, economic, and institutional aspects, not just biological aspects. This requires the involvement of multidisciplinary experts such as economists, social scientists, and policymakers to ensure diverse perspectives and interests are fully considered.

Secondly, appropriate governance mechanisms should be implemented to facilitate interdisciplinary collaboration and decision making. This may involve modifying existing fisheries management systems to integrate and balance all relevant considerations. Additionally, a transparent and participatory decision-making process should be adopted, encouraging all stakeholders to actively participate and provide their views and suggestions.

Furthermore, it is crucial to establish monitoring and evaluation mechanisms to track and assess the effectiveness of fisheries management and decision making. This will allow for necessary adjustments and improvements as needed.

In conclusion, achieving social, economic, and institutional integration is crucial to addressing the current biases and imbalances in fishery resource assessment and management. This requires involving multiple disciplines, establishing effective governance mechanisms, implementing transparent decision-making processes, and establishing robust monitoring and evaluation systems. By integrating these considerations, sustainable fisheries management goals can be achieved, leading to better outcomes and benefits for fishers, decision makers, and society as a whole.

5.2. Establish a Co-Management Mechanism of Fishery Resources Based on Fishing Community

The governance of SSFs is complex due to its characteristics of complexity, variability, publicity, and contradiction. Relying solely on the public sector to achieve positive governance results is often challenging, while the private sector, especially in the management of public resources, generally requires state support.

From the perspective of interactive governance theory, the logic of multi-subject collaborative governance aims to address the incompatibility between the traditional public management paradigm and the current times. The "Opinions of the Central Committee of the Communist Party of China and the State Council on Strengthening the Modernization of the Grassroots Governance System and Governance Capacity" issued in July 2021 emphasizes the importance of grassroots governance as the cornerstone of national governance [30]. Additionally, Central Document No. 1 in 2022 highlights the need to strengthen the construction of rural grassroots organizations and village party committees [31]. These documents provide a framework for China's rural governance system.

In the context of SSFs governance in China, the primary responsibility lies with the grassroots government at the county level and below. While the grassroots government is at the end of the administrative chain, it serves as the ultimate implementer and an important subject in the implementation of policies and government governance systems. It acts as the connection between the state and society and directly impacts the effectiveness of government policy implementation.

Therefore, the governance of SSFs should be primarily led by grassroots governance, specifically the government at the county level and below. However, the government should not be the sole independent governance body. It is crucial to involve more participants in small-scale fisheries governance. China's proposal to realize the modernization of national governance is based on the recognition of the limitations of a government-based and single national governance subject. From a pluralistic governance perspective, China's multi-subject collaborative governance should adhere to coordinated governance among multiple subjects under the leadership of the Communist Party of China, promoting governance consensus among all relevant parties.

In 2017, China issued a notice to strengthen the control of domestic fishing vessels and implement the total management of marine fishery resources. This notice introduced the management of fishing vessels based on classification and zoning. It stipulated that the indicators for small fishing vessels and their net tools should be approved and issued by the people's governments at the provincial level, with boats below 12 m being primarily managed by township regulations [16]. The township government, as the lowest level of the national administrative system, faces constraints from the top-down administrative system while also addressing the demands of rural society from the bottom-up. Therefore, a high degree of autonomy should be granted to township grassroots governance. This approach not only resolves the challenge of managing many small and scattered small-scale fisheries but also supports management reforms related to the total management of coastal fishery resources, quota fishing, designated landing points for catches, and catch traceability systems. It further strengthens the supervision of fishery production safety. Since 2017, China has implemented pilot projects on the management of marine fishery quota fishing, with varying degrees of fishery management organization for medium and large fishing vessels and certain fish species. However, the practical results of these pilots also face numerous challenges [32,33].

Therefore, in the current fishery transformation and reform in China, the governance system for small-scale fisheries should gradually explore the establishment of a governance pattern that incorporates grassroots party organizations representing national governance, administrative governance represented by townships, social governance represented by villagers' committees, scientists, fishery industry organizations, non-governmental organizations, and market governance represented by cooperatives.

5.3. Explore Mixed Governance Models

In the model of small-scale fisheries, hierarchical governance, co-governance, and autonomy are usually adopted. Hierarchical governance is the management of the state based on political preferences, and its policy objectives are achieved through centralized management and intervention (refer to Table 3). In the selection of small-scale fishery governance models, hierarchical governance dominates [34]. In Portugal, the octopus fisheries are governed using this model, which is top-down and combined with input and output control management measures. However, this model is usually more complex in practice than it is in theory, and it shows obvious shortcomings and dysfunction in the face of complex, multi-jurisdictional, and mixed fisheries, which are difficult to solve by relying on government departments alone [34,35].

Co-governance is different from hierarchical governance in that it emphasizes the sharing of power and responsibility among multiple parties (refer to Table 3). Due to the diversity of forms and rich connotations of joint governance [36], representative such as Japan's common fisheries management relies on two representative systems, FCAs (FCAs) and TURFs (Territorial Use Rights Fisheries, TURF) [37,38].

At the heart of the system is the definition of fishery resources as quasi-public goods, giving them a competitive and non-exclusive character. In France, the co-governance of small-scale fisheries mainly comes from the participation of stakeholders such as producer organizations, governments, markets, fishers, etc. Producer Originations (POs) play an important role in management, drafting regional fisheries management plans, and allocating quotas to the organization's members [35].

Autonomy reflects the situation in which actors take care of their own interests outside the purview of the government, and autonomy is not a capacity created by the government but is formed spontaneously [38,39]. FAO (Food and Agriculthre Organization) autonomy refers to the fact that fishery actors make their own governance decisions [40,41]. The model of self-government adopted in Chwaka Bay, Tanzania, is completely controlled by local elites and does not ensure the impartial, democratic representation, participation, and identity of village-level fishermen's committees, resulting in inefficient governance [34]. However, autonomy has played a positive role in Korea, which is based on fishing communities and relies on vertical cooperation between central and local governments, as well as cooperation between local governments and fishermen and interest groups, which are prerequisites for successful governance [39].

Mix governance emphasizes the participation of stakeholders at different stages of the decision-making process and plays a greater role than ever in the governance of public affairs, in contrast to the centrally authoritative nature of the traditional, hierarchical state, thereby improving the quality and effectiveness of governance [42]. This model not only represents the mixing and interaction between governance models but also emphasizes that the three types appear in different manifestations, proportions, or combinations. Emphasis is also placed on the interaction between the state, the market, and society [36].

Jentoft et al. argue that fisheries governance is a typical pyramid system with clear boundaries and information transmission chains between hierarchies, which is clearly different from other systems [43]. He once used "roses and pyramids" as metaphors for fishery governance. The "rose" symbolizes the system formed by the alliance of stakeholders, and the groups involved in decision-making and policy formulation jointly or interchangeably become the subject or object of governance, and they prefer to achieve tangible (quota) or intangible (political stance) support through active participation. It is an ongoing process of conflict resolution [42]. The above two management systems are based on two completely different management visions; that is, the concept and principles of meta-governance, as well as the corresponding system design and social interaction, should also be different.

In fact, the diverse, complex, and dynamic nature of SSFs is at odds with a single approach to governance [34]. Most of the time, the governance of SSFs should emerge in a more complex hybrid model and evolve over time as an adaptation to changing political, economic, or ecological environments [34]. The importance of SSFs lies not in their scale per se but in their diversity and complexity in many social, cultural, and institutional dimensions, which involve natural, social, and political dimensions rather than simply technical or scientific activities. Therefore, moving towards interactive governance is an inevitable choice for China's SSFs governance model.

Table 3. The Contents and Characteristics of Different Governance Modes.

Governance Model	Connotation	Characteristics
Hierarchical governance	Based on political preferences, top-down, centralized management and intervention to achieve policy objectives [36]	Centralized, command-and-control
Co-governance	Final decision making is delegated to all partners, with the government and stakeholders sharing power and responsibilities [36]	Polycentricity and pluralism of stakeholders
Self-governance	Autonomy refers to the fact that the fishery actors themselves make governance decisions [39,41]	spontaneity
Mixed governance	When faced with the governance problem of public affairs, a variety of governance decisions, schemes, styles, or models can be used to solve the problem [23,34,42,43]	Dynamic, adaptable, diverse

5.4. Use Policy Diverse Tools for Management

In order to improve the management and sustainable development of small-scale fisheries, a mixed policy approach should be adopted. Among them, input control is mainly used as the main management tool, and the form of output control and exclusive fisheries management areas (TURFs) are tried [5,38].

Traditionally, small-scale fisheries management has relied primarily on input control, i.e., limiting the resource input required for fishing activities, such as permits, quotas, and seasonal or regional restrictions. These measures aim to control fishing efforts and catch in order to maintain the sustainable use of fishery resources. However, input control alone may not solve all the problems facing fisheries management.

Therefore, a mix of policy instruments, such as output control and exclusive fisheries management areas (TURFs), should be tried [44,45]. Output control is the management of resources based on the output of fisheries, such as setting catch caps or limiting the number of individuals of a particular species or size that can be caught in a fishery.

This approach allows for more direct protection and maintenance of fishery resources and promotes their sustainable development. Specific sea areas are designated as exclusive fisheries management areas, which are jointly managed and protected by fishermen. This form of management can increase fishers' sense of responsibility and participation in the sustainable use of resources and promote the rational use of resources. Exclusive fisheries management areas can also promote community participation and democratization of fisheries governance, enhancing the sustainability of fisheries management.

SSFs can be managed more holistically through the use of a diverse policy approach, combining input control, output control, and exclusive fisheries management areas (TURFs). This diversified management approach aims to balance the sustainable use of fishery resources with the economic interests of fishers and to improve the effectiveness and sustainability of fisheries management. At the same time, a mixed policy approach can also increase the participation and consensus of fishers and relevant stakeholders and promote cooperation and joint efforts in fisheries management. This will contribute to the sustainable use of fishery resources, promote the economic benefits of fishers, and improve the effectiveness and sustainability of fisheries management.

5.5. Enhance Data Collection for SSFs to Facilitate Scientific Decision Making

The lack of reliable and comprehensive data has been a global and long-standing issue in SSFs [34,46,47]. This problem hinders the accurate assessment of the contributions and challenges of SSFs. Additionally, we must acknowledge that the characteristics of SSFs themselves make data collection difficult.

Different methods of data collection should be employed for different types of fisheries. In small-scale fisheries, maintaining fishing logs can be a challenging task. Therefore, we can ask fishermen to report data such as catch, catch volume, fishing areas, and trip durations to the fishing community upon their return to the fishing port. Furthermore, data regarding small-scale fisheries, including information on fishing vessels, fishing gear, practitioners, sales and distribution of catches, costs, and outputs, can be collected through monthly, quarterly, and annual surveys conducted by fishermen organizations. We can establish a database for these data and ensure its dynamic maintenance. Additionally, video monitoring devices can be used to gather data and enable real-time monitoring.

By employing these methods, we can gradually address the data collection challenges in small-scale fisheries, provide more accurate and comprehensive data support for scientific decision-making, and promote the sustainable development of SSFs.

6. Conclusions

The management of small-scale coastal fisheries is an area with multiple economic, social, and ecological objectives. China promotes the orderly management of coastal fisheries through a fishing licensing system. However, due to the multiple internal and external contradictions of fisheries, focusing only on a single value dimension, management mode, and means cannot alleviate the decline in nearshore fishery resources but poses a challenge to the resources and environment on which traditional fishermen rely for long-term survival. Although China's governance policies for small-scale coastal fisheries are improving, they still face challenges and limitations. With the development of other coastal industries, higher requirements are put forward for the management of coastal fisheries. The government should gradually raise the awareness of sustainable coastal fisheries, extend the policy focus from focusing on ecological values to social and economic values, enrich the governance subjects, adopt flexible governance models, increase the diversity of policy instruments, and improve the governance efficiency of coastal small-scale fishing fisheries.

Through continuous exploration and reform, the Chinese government is moving towards differentiated, refined, and scientific fishery management, which provides new opportunities for improving the management of SSFs in China. However, government knowledge of SSFs is still limited, and research and understanding of them need to be strengthened, as well as more comprehensive policy support measures to promote stakeholder engagement. Only in this way can the challenges of sustainable development of small-scale fisheries be addressed more effectively.

Under the framework of the Sustainable Development Goals (SDGs), the sustainability of fisheries is broader, covering not only the concept of sustainable fisheries but also the sustainable development of the entire fishery system, including the sustainability of the fishery industry chain, the sustainable development of fishing village communities, the livelihood of the fishery workforce, and the sustainability of fishery management policies.

Therefore, China's coastal sustainable fisheries policy should focus more on economic, social, and environmental considerations. This requires the government to gradually raise its understanding of sustainable coastal fisheries and adopt a variety of measures, including enriching the value of policies, increasing the number of governance subjects, and flexibly using governance models and diversified policy instruments to improve the governance efficiency of SSFs.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/fishes9110451/s1, Table S1: Policies names, release years, and references.

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References

1. Philip, A.L.; David, V.; Fazzino, I.I.; Melinda, A.; Ratana, C.; Glenna, G.; Moenieba, I. Fish and Food Security in Small-Scale Fisheries. In *Transdisciplinarity for Small-Scale Fisheries Governance. Analysis and Practice*; MARE Publication Series, 21; Chuenpagdee, R., Jentoft, S., Eds.; Springer Nature Switzerland AG: Cham, Switzerland, 2019. [CrossRef]
2. Food and Agriculture Organization of the United Nations. *Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries in the Context of Food Security and Poverty Eradication*; FAO: Rome, Italy, 2015; Available online: https://openknowledge.fao.org/items/6090777d-ded2-4c7c-9f00-a032fdc2690f (accessed on 27 September 2024).
3. Robert, I.A.; Daniel, J.S.; Anna, S.; Naazia, E.; Richard, M.F.; Rashid, S. Small-Scale Fisheries and Local Food Systems: Transformations, Threats and Opportunities. *Fish Fish.* **2022**, *23*, 109–124. [CrossRef]
4. Julia, N.; Ratana, C.; Svein, J. *Implementation of the Small-Scale Fisheries Guidelines a Legal and Policy Scan*; Springer: Cham Switzerland, 2024. [CrossRef]
5. Xiong, M.; Wu, Z.; Tang, Y.; Su, S.; Shen, H. Characteristics of Small-Scale Coastal Fisheries in China and Suggested Improvements in Management Strategies: A Case Study from Shengsi County in Zhejiang Province. *Front. Mar. Sci* **2022**, *9*, 811382. [CrossRef]
6. MORA. *Provisions on the Administration of Fishing Licenses*; Ministry of Agriculture and Rural Affairs: Beijing, China, 2019. Available online: http://www.yyj.moa.gov.cn/gzdt/201904/t20190418_6196078.htm (accessed on 24 October 2024).
7. MARA. *The 14Th Five-Year Plan for National Fishery Development Issued*; Ministry of Agriculture and Rural Affairs: Beijing, China, 2022. Available online: http://english.moa.gov.cn/news_522/202201/t20220110_300772.html#:~:text=The%20Ministry%20of%20Agriculture%20and%20Rural (accessed on 27 December 2022).
8. Satu, E.; Helvi, K. The Qualitative Content Analysis Process. *Adv. Nurs.* **2008**, *1*, 107–115. [CrossRef]
9. Wang, L.J. Research on the Transformation of Island Fishing Villages Based on the Investigation of Daishan Nanfeng Fishing Village. Master's Thesis, Zhejiang Ocean University, Zhoushan, China, 2020. (In Chinese).
10. Ping, Y.; Liu, D.D. 70 Years of Exploration on the Organization of Fishery Production and Management in China. *Chin. Rural Econ.* **2019**, *11*, 16–31. (In Chinese)
11. Lin, G.J. Fishery right system: 40 years' exploration of China's fishery economic institutional reform. *Chin. Fish. Econ.* **2019**, *37*, 114–125. (In Chinese)
12. Cheng, Y.F. Research on Double Control System of Fishing Vessel in China. Master's Thesis, Zhejiang Ocean University, Zhoushan, China, 2013. (In Chinese).
13. MARA. *Fishing "Zero Growth": A Harbinger of Structural Reform on the Fishery Supply Side*; Ministry of Agriculture and Rural Affairs: Beijing, China, 2019. Available online: http://www.yyj.moa.gov.cn/gzdt/201910/t20191016_6330010.htm. (accessed on 27 September 2024).
14. Zhao, L.L. Research on China's Community-Based Fisheries Management. Master's Thesis, Ocean University of China, Qingdao, China, 2009. (In Chinese).
15. Xiao, L. Chinese marine fishing vessel control has entered a new stage of "total reduction". *China Aquac.* **2004**, *1*, 8. (In Chinese)
16. MOA. *Notice of the Ministry of Agriculture on Further Strengthening the Management of Domestic Fishing Vessels and Implementing the Total Management of Marine Fishery Resources*; Ministry of Agriculture and Rural Affairs: Beijing, China, 2017. Available online: http://www.moa.gov.cn/nybgb/2017/derq/201712/t20171227_6130861.htm (accessed on 27 July 2024).
17. Shi, L.; Song, Y.N.; Qin, H. Changes, structure and optimization of China's marine fishing industry policy: A quantitative analysis based on policy text. *Mar. Sci.* **2021**, *45*, 40–50. (In Chinese)
18. Lorraine, M.M.; Richard, F.E. Getting the Job Done: Alternative Policy Instruments. *Educ. Eval. Policy. Anal.* **1987**, *2*, 133–152. (In Chinese)
19. Su, X.; Chen, X.J. Comparative Study for Input Control and Output Control in Fishery Management. *Trans. Oceanol. Limnol.* **2021**, *3*, 136–144. (In Chinese)
20. Bellido, J.M.; Sumaila, U.R.; Sánchez-Lizaso, J.L.; Palomares, M.L.; Pauly, D. Input Versus Output Controls as Instruments for Fisheries Management with a Focus on Mediterranean Fisheries. *Mar. Policy* **2020**, *118*, 103786. [CrossRef]
21. Qing, C. Policy Research on Fishery Resource Management in China. *Ocean Dev. Manag.* **2022**, *9*, 65–72. (In Chinese)
22. Wang, Y.N. Research on China's Offshore Fishing Management System-Based on "Input" and "Output" Control Perspective. Master's Thesis, Shanghai Ocean University, Shanghai, China, 2018. (In Chinese).
23. Svein, J.; Jahn, P.J. The Dynamics of Small-Scale Fisheries in Norway: From Adaptamentality to Governability. In *Interactive Governance for Small-Scale Fisheries: Global Reflections*; Springer: Cham, Switzerland, 2015; pp. 705–723. [CrossRef]
24. Cristian, A.; Johannes, G. Co-Management of Small-Scale Fisheries in Chile from a Network Governance Perspective. *Environments* **2020**, *7*, 104. [CrossRef]
25. Defeo, O.; Castrejon, M.; Perez-Castaneda, R.; Castilla, J.C.; Gutierrez, N.L.; Essington, T.E.; Folke, C. Co-Management in Latin American Small-Scale Shellfisheries: Assessment from Long-Term Case Studies. *Fish Fish.* **2016**, *17*, 176–192. [CrossRef]

26. Ana, C.E.D.; Ana, C.; Ana, M.P.; Cristiana, S.S. Participatory monitoring of small-scale coastal fisheries in South America: Use of fishers' knowledge and factors affecting participation. *Rev. Fish Biol. Fish.* **2020**, *30*, 313–333. [CrossRef]
27. Catherine, C.; Courtney, C. Thirty years after privatization: A survey of Icelandic small-boat fishermen. *Mar. Policy* **2017**, *80*, 69–80. [CrossRef]
28. Hu, W.; Liu, J.; Ma, Z.; Wang, Y.; Zhang, D.; Yu, W.; Chen, B. China's marine protected area system: Evolution, challenges, and new prospects. *Mar. Policy* **2020**, *115*, 103780. [CrossRef]
29. Carlos, A.R. Sustainability and Sustainable Development: A Review of Principles and Definitions. *Sci. Total Environ.* **2021**, *786*, 147481. [CrossRef]
30. Dong, L.R.; Gu, B.J.; Deng, Y.L. Suggestions on improving the modernization level of grassroots governance. *China Natl. Cond. Strength* **2022**, *4*, 24–27. (In Chinese)
31. CPC. *Opinion of the CPC Central Committee and the State Council on the Key Tasks for Comprehensive Rural Revitalization in 2022*; CPC Central Committee State Council: Beijing, China, 2022. Available online: http://www.gov.cn/xinwen/2022-02/22/content_5675035.htm (accessed on 22 March 2024).
32. Su, S.; Tang, Y.; Chang, B.; Zhu, W.; Chen, Y. Evolution of marine fisheries management in China from 1949 to 2019: How did China get here and where does China go next? *Fish Fish.* **2020**, *21*, 435–452. [CrossRef]
33. Kritzer, J.P.; Yi, T.; Yong, C.; Chris, C.; Sarah, G.; Tom, N.; Penas, E.; Keith, S.; Changchun, S.; Cody, S.; et al. Advancing multispecies fishery management in China: Lessons from international experience. *Aquac. Fish.* **2023**, *8*, 351–362. [CrossRef]
34. Pita, C.; Pereira, J.; Lourenço, S.; Sonderblohm, C.; Pierce, G.J. The Traditional Small-Scale Octopus Fishery in Portugal: Framing Its Governability. In *Interactive Governance for Small-Scale Fisheries*; MARE Publication, Series; Jentoft, S., Chuenpagdee, R., Eds.; Springer: Cham, Switzerland, 2015; Volume 13. [CrossRef]
35. Pascual-Fernández, J.; Pita, C.; Bavinck, M. (Eds.) *Small-Scale Fisheries in Europe: Status, Resilience and Governance*; MARE Publication Series; Springer: Cham, Switzerland, 2020; Volume 23. [CrossRef]
36. Kooiman, J. *Governing as Governance*; SAGE Publications: London, UK, 2003. [CrossRef]
37. Sakai, Y.; Matsui, T.; Yagi, N.; Senda, Y.; Kurokura, H. Econometric Analysis of the Factors Contributing to the Fish Price Increase in Coastal Turfs in Japan: The Case of Income-Pooling Fishery for Coastal Shrimp "Sakuraebi Sergia Lucens". *Fish. Sci.* **2010**, *76*, 711–718. [CrossRef]
38. Chi, N.T.Q.; Steven, S.; Atakelty, H.; Sayed, I. Territorial Use Rights for Fisheries (Turfs): State of the Art and the Road Ahead. *Mar. Policy* **2017**, *75*, 41–52. [CrossRef]
39. Sang-Go, L.; Amaj, R.M. Fishery Self-Governance in Fishing Communities of South Korea. *Mar. Policy* **2015**, *53*, 27–32. [CrossRef]
40. Lewis, P. Elinor's Ostrom's 'realist orientation': An investigation of the ontological commitments of her analysis of the possibility of self-governance. *J. Econ. Behav. Organ.* **2021**, *189*, 623–636. [CrossRef]
41. Food and Agriculture Organization of the United Nations. *Case Studies in Fisheries Self-Governance*; FAO: Rome, Italy, 2008; Available online: https://www.fao.org/4/a1497e/a1497e00.htm (accessed on 28 July 2024).
42. Leizhen, Z. Comparisons and Trends in Types of Governance: Commonality & Diversity. *J. Public Manag.* **2011**, *4*, 40–49.
43. Jentoft, S.; Chuenpagdee, R.; Bundy, A.; Mahon, R. Pyramids and Roses: Alternative Images for the Governance of Fisheries Systems. *Mar. Policy* **2010**, *34*, 1315–1321. [CrossRef]
44. Fernandez, M.; Kriegl, M.; Garmendia, V.; Aguilar, A.; Dulce, S.M. Evidence of illegal catch in the benthic artisanal fisheries of central Chile: Patterns across species and management regimes. *Lat. Am. J. Aquat. Res.* **2020**, *48*, 287–303. [CrossRef]
45. Wilen, J.E.; Cancino, J.; Uchida, H. The Economics of Territorial Use Rights Fisheries, or TURFs. *Rev. Environ. Econ. Policy* **2012**, *2*, 237–257. [CrossRef]
46. Thorpe, J.; McGoodwin, J.R. Crisis in the World's Fisheries: People, Problems and Policies. *J. Anim. Ecol.* **1996**, *65*, 132. [CrossRef]
47. Zeller, D.; Booth, S.; Davis, G.; Pauly, D. Re-Estimation of Small-Scale Fishery Catches for U.S. Flag-Associated Island Areas in the Western Pacific: The Last 50 Years. *Fish Bull.* **2007**, *105*, 266–277.

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