



UNIVERSITAT DE  
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## Addressing the reliability of data-poor stock assessment methods to provide advice on the status of small-scale fisheries

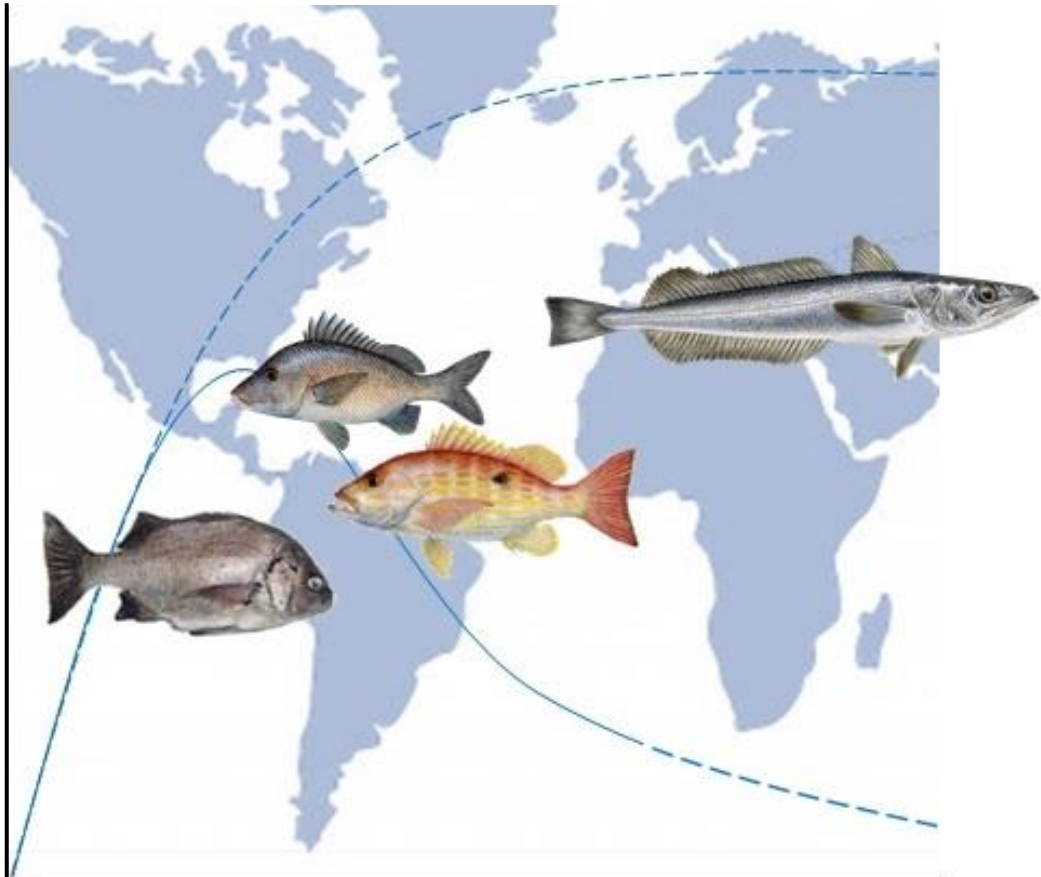
John Gabriel Ramírez Téllez

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Addressing the reliability of data-poor stock assessment methods to provide advice on the status of small-scale fisheries



**John Gabriel Ramírez Téllez**  
**Ph.D. Thesis**

Cover: pictures of fishes were modified from <http://fishbiosystem.ru> (*Haemulon plumierii*, white grunt (Eng)), [www.shrimpnfishflorida.com](http://www.shrimpnfishflorida.com) (*Lutjanus synagris*, lane snapper (Eng)), <http://www.conxemar.com> (*Merluccius merluccius*, European hake (Eng)) and <http://tumi.lamolina.edu.pe> (*Anisotremus scapularis*, Peruvian grunt (Eng)). Lower panel shows two Wayuu fishers who are recovering gillnets (lane snappers are caught) in the Colombian Caribbean Sea. (modified from Fundación Ecosfera).





# Addressing the reliability of data-poor stock assessment methods to provide advice on the status of small-scale fisheries

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# PROJECT FRAMEWORK

This thesis used four case studies to provide a wide view of the stock assessment issues for small-scale fisheries. Data from the European hake (*Merluccius merluccius*) fishery in GSA 06 (Spain) used in chapter 2.1 was supported by the EU H2020 Research and Innovation Action MINOUW (contr. nº 634495) project and EU Tender “DrumFish” Approaches to management for data-poor stocks in mixed fisheries (contract EASME/EMFF/2014/1.3.2.4/SI2.721116). Data from the European hake fishery in GSA 01 (Spain) used in chapter 2.2 was supported by the Fisheries General Directorate of the Murcia Region. The research of this chapter has received funding from the European Commission’s Horizon 2020 Research and Innovation Programme under Grant Agreement No. 773713 (“PANDORA”). Data from the Wayuu fishery (Colombian Caribbean) used in the chapter 2.3 was provided by the participatory monitoring program of artisanal fishery in the central Guajira that was supported by Fundación Ecosfera (Colombia) and Ecopetrol-Chevron Petroleum Company (Colombia). Finally, data from the beach fishery of Peruvian grunt used in chapter 2.4 was supported by the Monitoring and Assessment Program implemented by the Center for Conservation and Sustainability of the Smithsonian Conservation Biology Institute (PERU LNG and the Smithsonian Conservation Biology Institute Biodiversity, Peru and United States).

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Fisheries  
Bioeconomic  
Modelling



*To Martina*



# AKNOWLEDGMENTS

More than 10 years ago I had the opportunity to live and work for four years with the Amerindian Wayuu community in the northern Colombian Caribbean. This community lives in rural settlements, surrounded by a desert landscape, in houses made from cactus and mud, without access to utilities and getting water literally from the sky (using rain reservoirs). In other words, even today it is the place where I can come back to a far past as many things occur as they occurred 500 years ago. The economy and food supply of the Wayuu people inhabiting coastal areas relies on fisheries. When I finally left the Wayuu *rancherías*, I brought a lot of experiences I never thought to have but also I realized that other humans, other Colombian people, people from a developing country, fishers surviving in a remote place from my birth country daily face several problems finding food for them and for their families. Never again I was unable to stop seeing them. My largest acknowledgment is to them, because this thesis and the road travelled simply would not have existed without Wayuu fishers from Manaure and Uribia (La Guajira, Colombia). Especially thanks to my colleague, Gina Puentes who the Wayuu-fishery adventure started with. Thanks to Francisco Reyes, Luis Rodríguez, Miller Camargo and all Fundación Ecosfera team for the invaluable work to help improving the Wayuu fisheries. Thanks to all Wayuu people and Fundación Ecosfera that lead the largest participatory monitoring program of marine fisheries in Colombia (since 2006) because it was the seed of this thesis.

My days working with Wayuu fishers taught me the importance to properly determine the status of the resources harvested by fisheries if we wish to contribute to improve the livelihoods of artisanal fishers while resources are healthy exploited. Accordingly, I was involved in the scientific department of the national fishery authority of Colombia (AUNAP). There, I discovered that one of the reasons why Colombia does not care enough about the status of fishery resources is because the lack of knowledge of life-history of target species and partial understanding of hundreds of artisanal fisheries look like a huge wall to be climbed. Consequently, the stock assessment tasks are performed paying insufficient attention to assumptions and caveats of models. In other words, the availability of few and biased data as well as misunderstanding of the

## Acknowledgement

models used to define catch limits or stock status lead to a hopeless picture. The former director of the knowledge office of the AUNAP, Vladimir Puentes, promoted me to go deeper in the participatory perspective for monitoring and managing artisanal fisheries, which is highly desirable when catch history is not available. Thanks to AUNAP and Vladimir for all your support. This was the period when I identified that my PhD thesis should be focused on improve the way to advice artisanal fisheries (small-scale fisheries).

After landing in Spain, I worked with Ferran Pereira (rest in peace) who was a passionate person in small-scale fishers in Latin America. He permeated me with the love for fishery science and we spent a really good time together while my M.Sc. thesis was getting shape. By the way, it was the first approach to the understanding of the Wayuu fisheries. Additionally, Ferran brought me the opportunity of developing the monitoring program of fish and fisheries associated with a liquefaction plant of gas in the central coast of Peru. That experience allowed me to expand my view on artisanal fisheries, also incorporated in this thesis. For all the above reasons, you are an important part of this thesis wherever you are. Thanks to Mikel Zabala for his valuable suggestions during my M.Sc. studies.

Then, I started to work with Jordi Lleonart my Ph.D. advisor during the first years before he becomes retired. I would like to mention a particular moment that changed the focus of my thesis. While I was defining the input data to perform the pseudo-cohort VPA of the lane snapper (Wayuu fishery), I asked him: which of the von Bertalanffy growth parameters sets that I have estimated is the correct one to define the growth of this species?'. His answer was forceful: "I don't know". I remember that his answer was very frustrating to me but it also was the motivation to explore what exactly means an "I don't know" in terms of stock assessment of small-scale fisheries. This thesis tilts around this question. Additionally, I thank Jordi for introducing me to the stock assessment world of "data-rich" fisheries in the Mediterranean. Through participating in several stock assessment meetings, I realized that knowledge of data-rich and data-moderate stock assessment is highly valuable to recognize the caveats and scope of data-limited/poor fisheries.

## Acknowledgement

Developing my thesis I had the opportunity of work in several European projects. Thanks to Paloma Martín for involving me in the DrumFish project that helped me to develop my skills analyzing data-poor Catalan fisheries. Additionally, thanks to Laura Recasens to provided me with support and data to go deeper in the European hake fisheries in the Northwestern Mediterranean. This species was the “guinea pig” that allowed me to test the answer of data-poor models to different assumptions and data availability. A special acknowledgment to Jason Cope, for providing a kind and strong advice to go deeper in the stock assessment of data-limited stocks, but also for spending time in his laboratory in Seattle (USA) discussing our passion: to help people from the data-limited stock-assessment arena.

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# RESUM

Els pescadors artesanals s'identifiquen generalment com actors claus en la recuperació de poblacions sobreexplotades, en el subministrament d'aliment a poblacions rurals i en contribuir en l'obtenció d'ecosistemes marins saludables. Les poblacions que són extretes per les pesqueries artesanals (denominades pesqueries de petita escala) tendeixen a manca d'avaluació. No obstant, alguns mètodes d'avaluació que s'inclouen en el grup de 'dades limitades' ajuden a subministrar informació sobre l'estat d'aquestes pesqueries quan no posseeixen informació històrica de captures. Molts dels mètodes limitats en dades i basats en estructura de talles assumeixen estat d'equilibri. Aquests mètodes utilitzen al menys com dades d'entrada l'estructura de talles que provenen de la pesqueria i el coneixement disponible sobre els paràmetres de història de vida de les espècies pescades. Per tot lo exposat anteriorment, el propòsit d'aquesta tesi és direccionar la fiabilitat dels mètodes limitats en dades per proporcionar informació sobre l'estat de les poblacions que no compten amb la informació històrica de captures. Les pesqueries de lluç (*Merluccius merluccius*) d'arrossegament en GSA 06 (Mar Mediterrani nord-oest) 'rica en dades' va ser assumida com limitada en dades. Aquest cas d'estudi em va permetre provar l'acompliment de l'Anàlisi de Població Virtual (VPA per les seves sigles en anglès) basat en pseudo cohorts quan les dades d'entrada s'assumeixen ben conegudes i lliures de biaixos. Aquesta mateixa pesqueria però conduïda en GSA 01 (sud-oest del Mar Mediterrani) també va ser utilitzada com un estudi de cas limitat en dades. Això amb el propòsit d'explorar la incertesa produïda pel model que determina la proporció del potencial reproductor que és basada en talles (LB-SPR per les seves sigles en anglès) quan són considerades dues hipòtesis de creixement contrastants. Reconeixent els desafiaments que enfronta l'avaluació de les pesqueries artesanals al voltant del món, vaig considerar l'anàlisi de biaixos rellevants en les dades que alimenten el model així com una gran incertesa en els resultats d'avaluació. D'aquesta manera, l'efecte dels biaixos en la informació provinent de les pesqueries i la incertesa en els paràmetres de història de vida sobre els resultats del VPA basat en pseudo cohorts es va explorar a través de l'avaluació de les pesqueries Wayuu del 'pargo rayado' (*Lutjanus synagris*) i la 'boca colorada' (*Haemulon plumierii*) al nord del Mar Carib colombià. Posteriorment, un extrem però cas comú d'incertesa en les pesqueries artesanals es va considerar a través de l'avaluació de la pesqueria de platja de la 'chita' (*Anisotremus scapularis*) en la costa central de Perú en l'oceà Pacífic. Aquesta pesqueria conté informació limitada sobre la història de vida de l'espècie i sobre les captures produïdes per la pesqueria. Els meus resultats indiquen que el VPA basat en pseudo cohorts pot oferir informació útil respecte la tendència d'explotació de la pesqueria, però els valors absoluts no expressen apropiadament la mortalitat per pesca ni per la mida de la població entre anys pel lluç. El valor de SPR per aquesta mateixa espècie no es vincula a la hipòtesi de creixement assumida i les estimacions de la proporció de mortalitat per pesca respecte a la mortalitat natural ( $F/M$ ) y el valor de SPR depenen de la mida de la mostra i de la representació de l'estructura de talles de la població pescada. La contribució de la informació derivada del monitoratge participatiu de la pesca artesanal a Colòmbia demostra una millora de la imatge d'explotació del 'pargo rayado' i la 'boca colorada' que no quan només s'utilitza la informació oficial. D'altra banda, vaig trobar que la incertesa relacionada amb els paràmetres de creixement de von Bertalanffy i la mortalitat natural de la 'chita' pot ser dirigida però una definició precisa del valor SPR no s'obté fàcilment. Aquesta tesi ressalta que els mètodes limitats en dades que assumeixen un estat d'equilibri poden contribuir a definir l'estat d'explotació de les pesqueries artesanals. No obstant, l'estat de la població es troba afectat de manera important pels biaixos de les dades que alimenten el model, el coneixement disponible sobre la pesqueria avaluada i el com la pesqueria s'ajusta als supòsits dels models d'avaluació.



# RESUMEN

Los pescadores artesanales se identifican generalmente como actores claves en la recuperación de poblaciones sobreexplotadas, en el suministro de alimento a poblaciones rurales y en contribuir a alcanzar ecosistemas marinos saludables. Las poblaciones que son extraídas por las pesquerías artesanales (denominadas pesquerías de pequeña escala) tienden a carecer de evaluación. Sin embargo, algunos métodos de evaluación que se incluyen en el grupo “datos limitados” ayudan a suministrar información sobre el estado de estas pesquerías cuando no poseen información histórica de capturas. Muchos de los métodos limitados en datos y basados en estructura de tallas asumen estado de equilibrio. Estos métodos usan al menos como datos de entrada la estructura de tallas que provienen de la pesquería y el conocimiento disponible sobre los parámetros de historia de vida de las especies pescadas. Por todo lo expuesto anteriormente, el propósito de esta tesis es direccionar la fiabilidad de los métodos limitados en datos para proporcionar información sobre el estado de poblaciones que no cuentan con información histórica de capturas. La pesquería de merluza (*Merluccius merluccius*) de arrastre en GSA 06 (Mar Mediterráneo noroeste) “rica en datos” se asumió limitada en datos. Este caso de estudio me permitió probar el desempeño del Análisis de Población Virtual (VPA por sus siglas en inglés) basado en pseudo cohortes cuando los datos de entrada se asumen bien conocidos y libres de sesgos. Esta misma pesquería pero llevada a cabo en GSA 01 (suroeste del Mar Mediterraneo) también fue usada como un estudio de caso limitado en datos. Esto con el propósito de explorar la incertidumbre producida por el modelo que determina la proporción del potencial reproductor que es basada en tallas (LB-SPR por sus siglas en inglés) cuando son consideradas dos hipótesis de crecimiento contrastantes. Reconociendo los desafíos que enfrenta la evaluación de las pesquerías artesanales alrededor del mundo, yo consideré el análisis de sesgos relevantes en los datos que alimentan el modelo así como una gran incertidumbre en los resultados de evaluación. De esta manera, el efecto de los sesgos en la información proveniente de las pesquerías y la incertidumbre en los parámetros de historia de vida sobre los resultados del VPA basado en pseudo cohortes se exploró a través de la evaluación de las pesquerías Wayuu del pargo rayado (*Lutjanus synagris*) y la boca colorada (*Haemulon plumieri*) en el norte del Mar Caribe colombiano. Posteriormente, un extremo pero común caso de incertidumbre en pesquerías artesanales se consideró a través de la evaluación de la pesquería de playa de la chita (*Anisotremus scapularis*) en la costa central de Perú en el Océano Pacífico. Esta pesquería posee limitada información sobre la historia de vida de la especie y sobre las capturas producidas por la pesquería. Mis hallazgos indican que el VPA basado en pseudo-cohortes puede ofrecer información útil respecto a la tendencia de explotación de la pesquería, pero los valores absolutos no expresan apropiadamente la mortalidad por pesca ni el tamaño de la población entre años para la merluza. El valor de SPR para esta misma especie no se vincula a la hipótesis de crecimiento asumida y las estimaciones de la proporción de mortalidad por pesca respecto a la mortalidad natural ( $F/M$ ) y el valor de SPR dependen del tamaño de la muestra y de la representación de la estructura de tallas del stock. La contribución de la información derivada del monitoreo participativo de la pesca artesanal en Colombia demuestra mejoramiento de la imagen de explotación del pargo rayado y la boca colorada que no cuando solo se usa información oficial. De otro lado, encontré que la incertidumbre relacionada con los parámetros de crecimiento de von Bertalanffy y la mortalidad natural de la chita puede ser direccionada pero una definición precisa del valor de SPR no se obtiene fácilmente. Esta tesis resalta que los métodos limitados en datos que asumen estado de equilibrio pueden contribuir a definir el estado de explotación de las pesquerías artesanales. Sin embargo, el estado de la población se encuentra afectado de manera importante por los sesgos de los datos que alimentan el modelo, el conocimiento disponible sobre la pesquería evaluada y el cómo la pesquería se ajusta a los supuestos de los modelos de evaluación.



# ABSTRACT

Small-scale fishers are often identified as key players in the recovery of overexploited fish and invertebrate stocks supplying food for rural people and contributing to achieving healthy marine ecosystems. Stocks harvested by small-scale fisheries tend to be largely unassessed, but methods based on the data-limited toolbox exist that help provide information on exploitation status for fished stocks that do not have historical time series of catches, as usually occur in small-scale fisheries. Many of the data-limited methods follow length-based assessment approaches, which assume steady state, use at least length structure derived from fishery and knowledge on the life history parameters of the fished species. Accordingly, this thesis aimed at addressing the reliability of data-poor stock assessment methods in providing advice on the status of small-scale fisheries lacking knowledge of catch history. The data-rich bottom trawl fishery for European hake (*Merluccius merluccius*) in GSA 06 (Northwest Mediterranean Sea) was assumed as data-limited. This case study allowed me to test the performance of the pseudo-cohort Virtual Population Analysis (VPA) when input data are considered well known and unbiased. The same fishery but held in GSA 01 (Southwest Mediterranean Sea) was used as data-limited case study to introduce the uncertainty derived from parameterizing the length-based spawning potential ratio (LB-SPR) model with two contrasting growth hypotheses. Acknowledging challenges faced by stock assessment of small-scale fisheries around the world, I considered high input-data bias and large outputs uncertainty. The effect of biases in fishery data and uncertainty in life-history parameters on the outputs of the pseudo-cohort VPA model was explored by assessing the small-scale Wayuu fisheries for lane snapper (*Lutjanus synagris*) and white grunt (*Haemulon plumierii*) in the northern Colombian Caribbean Sea. An extreme, but common, case of uncertainty in small-scale fisheries was explored through assessing the beach fishery of the Peruvian grunt (*Anisotremus scapularis*) in the central coast of Peru on the Pacific Ocean, holding poor information on life-history parameters and catches. My findings indicate that the pseudo-cohort VPA may offer useful information regarding the exploitation trend but the absolute values of the indicators do not accurately express the fishing mortality and stock size among years for the European hake. The SPR estimates for this species is not specially linked to the growth hypothesis, and estimates of the ratio of fishing mortality to natural mortality ( $F/M$ ) and the SPR value depend on the sample size and representation of the stock structure. The contribution of the information derived from the participatory monitoring of small-scale fisheries in Colombia, instead of using only official fishery data, largely demonstrated an improving picture of the exploitation of the lane snapper and white grunt. The uncertainty related to estimates of the von Bertalanffy growth parameters and natural mortality of the Peruvian grunt could be addressed but an accurate definition of SPR was not straightforwardly achieved. This thesis highlights that the data-limited methods assuming a steady state might contribute to defining the status of the small-scale fisheries. However, the stock status is importantly affected by bias in the input data, the available knowledge on the assessed fisheries and how fishery fit the model assumptions.



# 1. INTRODUCTION



### 1.1. Assessment and status of marine fisheries

The fishery resources, and particularly those harvested in marine waters, are quite difficult to quantify. Answering key questions, including, how many fish do remain in the sea under the current fishing effort and selection patterns and how much biomass can be harvested without affecting the natural recovery of the stock is usually not straightforward. The early work of Hjort and Lea (1914), who followed the outstanding abundance peak of herrings spawned in 1904, suggested that survival of eggs may largely vary from year to year. This result helped to set foundations for the demography of fish (birth rate, age distribution and migration). Thus, age data were routinely incorporated into assessment of populations of fish since the 1900s (Jackson, 2007). Consequently, by observing the consistent progression of strong year classes while they aged, the status of fisheries was determined (Smith, 2002).

Ricard and colleagues (2011) examined the available knowledge on the status of commercially exploited marine species with the RAM Legacy Stock Assessment Database. These authors found represented only 16% of harvested fish taxa, which were mainly composed by stocks from Europe, North America, New Zealand, Australia, Russia, South Africa, Argentina (industrial fishery) and Peru (industrial fishery), Iran and from eight regional fisheries management organization focused mainly in tuna stocks. However, as observed above and mentioned by Hilborn and Ovando (2014), the stock assessments are mainly biased toward large, commercially important species harvested in developed countries. Additionally, Costello et al. (2012), Hilborn and Ovando (2014) and Garcia et al (2018) identified that the scientifically assessed stocks are in better shape (and frequently rebuilding) than unassessed small stocks, often in developing countries (and frequently depleting). Although several well-assessed stocks in developed countries are rebuilding (Fig. 1), most of the stocks harvested by the world fisheries (most of them in developing countries) are unassessed (Fig. 2) or are observed as overexploited (Pauly and Zeller, 2017a).

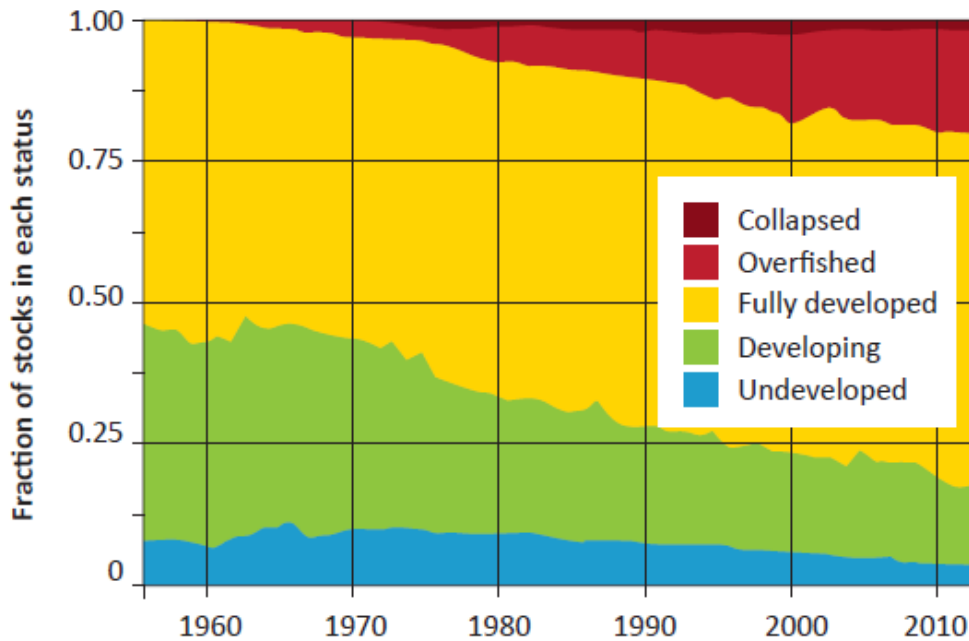


Fig. 1. Global trend of fisheries, including assessed and unassessed stocks. Modified from Garcia et al. (2018).

based on the Time trend of median Stock status in terms of the ratio of Biomass ( $B$ ) to the Biomass level that allows the maximum sustainable yield ( $msy$ ). This figure is taken from Costello et al. (2012).

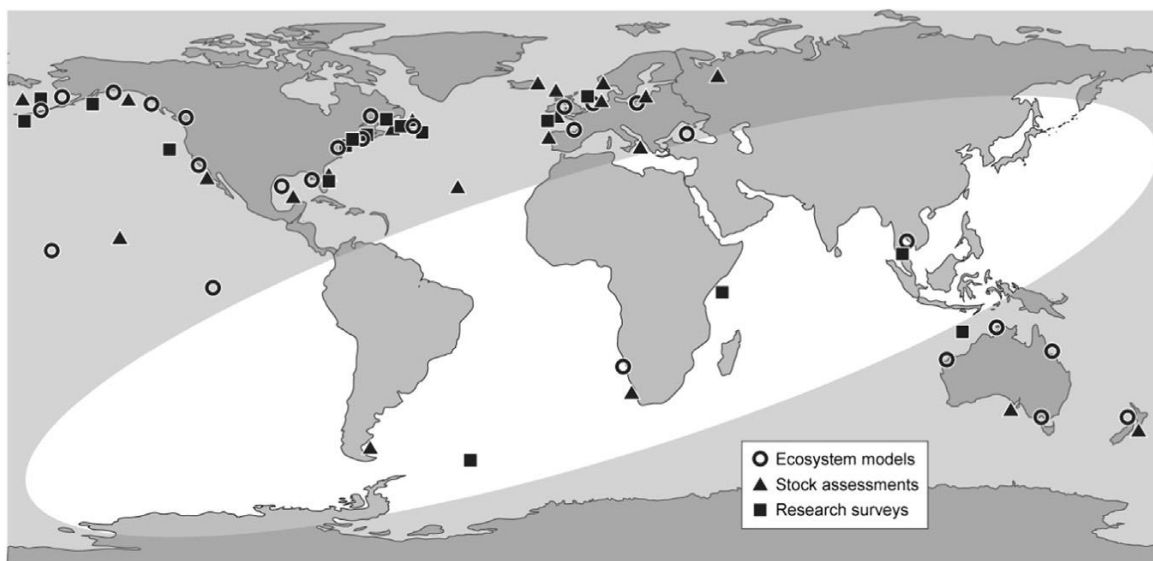


Fig. 2. Pauly and Zeller, (2017a) modified the version of the figure 1B performed by Worm et al., (2009), in order to demonstrate that the conclusion of “world” fisheries are rebuilding is biased due to most of stock assessment has been performed in developed countries. Most of resources harvested in South America, Africa and Asia, and the Mediterranean Sea, are unassessed.

This thesis section (1. Introduction), including the current subsection (1.1. Assessment and status of marine fisheries), deals with the main concepts and framework required to understand the challenges of performing stock assessment of

small-scale fisheries, mainly in developing countries. The subsection 1.2 (Assessing the status of small-scale fisheries: Colombian case) is dedicated to describe the motivation behind this thesis. Thus, this subsection is not concerned with the description of the stock assessment concepts (see other subsections) but depicts the limitations that may underlie the stock assessment of small-scale fisheries. Subsection 1.3 (Stock assessment according to data availability) explains what are the main criteria used by fishery science to determine whether a fishery or stock is categorized as “data-rich”, “data-moderate”, “data-limited” or “data-poor”. Depending on what category is attributed to a stock, the assessment is performed under one or other assumptions.

Subsection 1.4 (Uncertainty in the stock assessment outputs) is going beyond the impact of data availability on stock assessment to understand how the uncertainty related to data inputs affects the stock assessment results. In other words, how realistic and accurate is the status derived from stock assessment. Subsection 1.5 (Bias in data and information derived from small-scale fisheries) explains the limitations to use and interpret stock assessment models when data from small-scale fisheries is used to set up the assessment model. One particular subsection (1.6. “Robin Hood” approach) is dedicated to a relative recent alternative that fishery science has implemented to make the stock assessment of data-limited or data-poor resources feasible. This subsection is quite important because the data-poor resources have been benefited from implementing this approach that “borrows” data from other stocks (e.g. those taxonomically related) to set the stock assessment model for a particular species. However, this procedure may cause problems to interpret the stock assessment results. Finally, the subsection 1.7 (Research and structure of the thesis), present the main objective as well as the specific objectives of this thesis. Additionally, there is a brief explanation of the context in which the different chapters have been developed.

### 1.2. Assessing the status of small-scale fisheries: Colombian case

How to provide reliable information on the status of small-scale fisheries? This question was initially rooted in the fact that little is known about the status of the Colombian small-scale fisheries (SSF) (about SSF definition, see section 1.5) and there is a lack of knowledge of catch history and life-history parameters for most harvested species. In fact, the research methods to answer this question are relevant to many

other SSF in developing countries and this fishery has been used as a case-study. The Colombian fishery laws require setting annual catch limits (ACL) by enforcing “total allowable catch” (Ministry of Agriculture, 1991). Given that reliable time series of catches are unavailable, methods that either determine acceptable biological catch as a basis for setting ACL or are capable to provide stock status from catch history and some auxiliary information are not suitable (Carruthers et al., 2014; Froese et al., 2016; Pedersen and Berg, 2017).

Stock status and a global ACL for around 20 species in the Caribbean Sea and Ocean Pacific of Colombia have been defined using length-based methods (Barreto and Borda, 2008; Puentes et al., 2014). The stock assessment methods used by these authors assumed that recruitment ( $R$ ), fishing mortality ( $F$ ) and natural mortality ( $M$ ) remained constant with time (Sparre et al., 1989). These assumptions are required to perform stock assessment methods using data from one year in isolation, implying that a pseudo-cohort (i.e. number-at-age observed during a single year) may be used to describe a real cohort (number-at-age that remains from recruitment age to the maximum harvested age). Thus, the stock assessments are performed under a “constant parameter system” (Sparre et al., 1989), steady-state (Lleonart and Salat, 1997) or equilibrium conditions (Prince et al., 2015c). In the Colombian case, the  $F$  and number of fish that resulted from that pseudo-cohort analysis plus some life-history parameters were used for determining the biomass ( $B$ ) that may be harvested without exceeding the level of maximum sustainable yield ( $B_{msy}$ ) (Sparre et al., 1989). Estimates of  $B_{msy}$  was used to support recommendations of annual catch limit for the Colombian marine resources (Barreto and Borda, 2008; Puentes et al., 2014).

Although catch history was not used to provide information on stock status, the above-mentioned analyses require accurate knowledge of the growth curve. However, ageing methods based on otoliths readings of tropical fish face problems to interpret the relative ages because seasonal changes in water temperature are not large enough with increasing proximity to the equator (Newman et al., 2000). Additionally, tagging-recapture experiments are expensive and very low number of tagged individuals are usually recaptured. It is not usual to use this technique to estimate the age of commercial fish, even in developed countries, because there is very low success when

recapturing the marked individuals. For instance, only 1.3% of 527 tagged European hake off the northwest Iberian Peninsula were recaptured in a tagging experiment (Piñeiro et al., 2007). Therefore, estimates of the growth curve of tropical fishes are usually based on length methods (Gayanilo et al., 2005; Mildenerger et al., 2017; Taylor and Mildenerger, 2017). These methods are less expensive to implement and may produce higher suitable data to ageing fish than otolith readings and tagging recapture experiments. However, length-based method could be affected by several sources of bias, including population exhibiting age truncation (Kenchington, 2014; Saari et al., 2014), the underestimation of smallest and largest fishes (Then et al., 2015) and the misunderstanding of the sampling vulnerability schedule (Gwinn et al., 2010).

For instance, length-based methods and otoliths readings have been used to estimate the von Bertalanffy growth parameters of Lane snapper (*Lutjanus synagris*), a highly-valued demersal fish of the Wayuu fishery (Box 1) and one of the most studied demersal fish in the Caribbean Sea. Estimates of asymptotic length ( $L_{\infty}$ = 312-708 mm TL) and growth rate ( $k$ = 0.04-0.73 yr<sup>-1</sup>) along its distribution area involved large variability (Acosta and Appeldoorn, 1992; Barreto and Borda, 2008; Frédou et al., 2009; Garcia and Duarte, 2006; Gómez et al., 2001; Johnson et al., 1995; Leite Jr. et al., 2005; Luckhurst et al., 2000; Manickchand-Dass, 1987; Manjarrés, 2004; Puentes et al., 2014). Bias in estimates of these parameters or uncertainty related to the “true” value of  $L_{\infty}$  and  $k$  are propagated to  $M$  in the stock assessment analyses because most of the indirect estimators of natural mortality are based on these von Bertalanffy growth parameters (*vBGP*) (Abella et al., 1997; Gislason et al., 2010; Kenchington, 2014; Then et al., 2015). Wrong estimates of  $M$  may be quite problematic since it is recognized that  $M$  produces a high impact in the final stock assessment outputs (Gwinn et al., 2010; Hordyk et al., 2015; Kenchington, 2014; Kokkalis et al., 2017; Scott et al., 2016).

## Box 1. Small-scale Wayuu fishery (Chapter 2.3)

**Study area:** The Wayuu fishery is carried out by the Amerindian community Wayuu (Figure a). This community holds its own language, heritage territory and laws (Alarcón Puentes, 2006). The Wayuu community inhabit a desert landscape extended along the northern region of the Republic of Colombia and the Bolivarian Republic of Venezuela. Wayuu people lives in small village surrounded by desert and its economy is based on small-scale fisheries (Puentes et al., 2012), grazing of goats and sheep (Fuenmayor et al., 2004) and handmade of bags and hammocks (Iguaran and Rivadeneira, 2014). This community lives in poverty and more than 90% of population exhibit unsatisfied basic needs (DANE, 2005).



Figure a. People and territory Wayuu. Source of left (RegiónCaribe.org, 2018) and right pictures (agenciadenoticias, 2017), respectively.

In Colombia, the Wayuu community lives in the Department of La Guajira in the northern region of South America (Figure b). The Wayuu fishery is carried out during all year on a wide continental shelf that is enriched by an upwelling in the dry season (December-April) (Andrade and Barton, 2005).



Figure b. Study area.

**Fishery:** The Wayuu fishery is performed using wooden or fiber-glass boats (length size < 8 m) propelled by sail or internal engine (< 40 HP) (Manjarrés, 2004) (Figure c). The gears used to

fish include gillnets, encircling nets, traps, longlines, hand lines and spear guns (Puentes et al., 2012).



Figure c. Boats used by Wayuu fishery (Fundacion Ecosfera, 2009)

The last fishery census was held in 2000 and near 800 boats were fishing in coastal areas of La Guajira (Manjarrés, 2004). Eight species represented 80% of catches but the lane snapper (*Lutjanus synagris*) is the highest valued species and white grunt (*Haemulon plumierii*) is the most landed species (Gómez et al., 2004) (Figure d). This two species represented mean annual landings around 60 and 200 tons (2006-2011), respectively, in the central Guajira (Puentes et al., 2012).

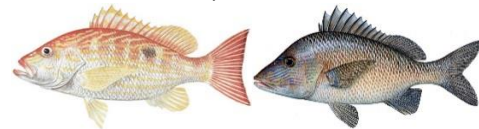


Figure d. Lane snapper (left, Fishidentification, 2010) and white grunt (right, FishIndex, 2010).

**Life-history parameters:** The von Bertalanffy growth parameters, natural mortality and size of maturity of lane snapper have been estimated around the Caribbean basin and also in Colombian Caribbean waters. However, these estimates may be merging several stocks and being based on biased input data (Explained and discussed in chapter 2.3). Life history parameters of white grunt have been never assessed in Colombian waters.

**Stock assessment:** Lane snapper in Colombia has been frequently assessed using length-based methods (Barreto and Borda, 2008; Puentes et al., 2014). The stock is considered as overexploited while the management has been weakly enforced. Most recent "stock" advice was based on fishery indicators while formal assessment was not carried out (Duarte et al., 2018). The white grunt stock has been never formally assessed.

Size of maturity of Lane snapper also varied along the Colombian Caribbean because more than one population was recorded at the basin scale (Landínez-García et al., 2009; Manjarrés, 2004). Selectivity of the fishing gears used by small-scale fisheries that sampled the fish to carry out reproductive studies affected the estimates of the size of maturity (García and Ramírez, 2016). This implies that size of maturity that is perceived by monitoring of small-scale fisheries may be biased, meaning that also fisheries, and not exclusively biological or environmental issues, affect its estimation. In fact, Prince et al. (2015) found that when the structure of the population is heavily truncated by overfishing, it is difficult to sample size classes with 100% of mature individuals. Accordingly, to achieve accurate estimates of life-history parameters of small-scale fishery resources (such as  $L_{\infty}$ ,  $k$ ,  $M$  and size of maturity) may be hindered by the process of gathering and analyzing fishery-dependent data.

Catch in number (and biomass) by length/age group is expected to be known when the stock assessment is used to provide an absolute estimate of the stock size and determine the stock portion that may be removed without causing overfishing (e.g. Annual Catch Limit). However, small-scale fisheries often use a wide range of landing sites along the coast, implying that the total biomass removed by all fleets could be difficult to estimate (Salas et al., 2007). While providing an accurate relative stock status trend may be still feasible if underestimation of catches is constant throughout the studied years, unequal record of landings among years may lead to a biased definition of the trend of stock status and exploitation level (Rudd and Branch, 2016). In order to overcome issues related to misspecification in stock size, it is common that status of fisheries is advised in relative terms (e.g.  $F/F_{msy}$ , spawning potential ratio or  $B/B_{msy}$ ) (Carruthers et al., 2014; Costello et al., 2012; Hilborn and Ovando, 2014; Hordyk et al., 2015). However, even relative definition of stock status may lead to misleading results when annual changes in fishing effort among landing points are not considered (Salas et al., 2007).

Recalling the above-enunciated question, a reliable definition of the status of stocks harvested by small-scale fisheries depend on data availability (Carruthers et al., 2014; Chrysafi and Kuparinen, 2015), uncertainty related to model input parameters

(Hordyk et al., 2015; Kokkalis et al., 2017; Thorson et al., 2017) and biases behind estimates of the model input data (Gwinn et al., 2010; Kenchington, 2014; Rudd and Branch, 2016; Salas et al., 2007). In the next section I introduce some of these aspects in depth offering the required context within which the stock assessment of small-scale fisheries is performed.

### 1.3. Stock assessment according to data availability

The stock assessment performed on small-scale Colombian resources was carried out in a context of limited or “poor” data availability. Therefore, it can be deduced that stock assessment quality will depend on the kind of data fed into the stock assessment model (the model hereinafter). Before differentiating among these data availability categories, it is appropriate to define the meaning of the word "status" in the stock assessment context. Status, or in a more precise way, biological status relies on the stock assessment from structural population models capable to estimate fishing mortality and stock size (Costello et al., 2012). Thus,  $F$  and certain measure of stock size (usually biomass,  $B$  or spawning stock biomass,  $SSB$ ) are compared against a particular value (threshold) that determine a change of status (e.g. from sustainably exploited to overexploited).  $B < B_{threshold}$  indicates that the stock status is overfished, while  $F > F_{threshold}$  is producing overfishing (Cooper, 2006, Table 1).

Table 1. A stock may be overexploited and/or overfishing may be occurring according to the levels of biomass ( $B$ ) and fishing mortality ( $F$ ) regarding the threshold. Modified from Cooper (2006). Traffic light colors indicate desirable (green), undesirable and should be improved (yellow) and problematic (red) stock conditions.

	$B < B_{threshold}$	$B \geq B_{threshold}$
$F \geq F_{threshold}$	Stock is overfished & Overfishing is occurring	Stock is not overfished but overfishing is occurring
$F < F_{threshold}$	Stock is overfished but overfishing is not occurring	Stock is not overfished & overfishing is not occurring

Fisheries that hold long time series of catch-at-age data (e.g. at least twice the lifespan of target species), have abundance indices (e.g. fishery-independent data from surveys), known fishing effort (e.g. number of days at sea per year), and where life-history parameters are properly estimated, are defined as data-rich (Chrysafi and

Kuparinen, 2015). The status of these fisheries can be assessed by sophisticated models, including statistical catch-at-age methods (Methot Jr and Wetzel, 2013) and those based on virtual population analysis (Shepherd, 1999). Using these data-rich methods, it is accepted that estimates of fishing mortality (harvesting) and stock size (e.g. number of individuals or stock spawning biomass) are unbiased, while the uncertainty of the outputs can be properly considered (Wetzel and Punt, 2015).

Although some authors include surplus production models as part of data-rich methods because long time series of catches, as well as abundance indices, are required (Carruthers et al., 2014; Froese et al., 2016; Pedersen and Berg, 2017), these methods are often used when catch-at-age is unknown. The “assessments for all” initiative (a4a) defines the data-moderate concept from the same information that is used by data-rich methods (Jardim et al., 2015b). Despite nominal effort not being required for this data-moderate model, probably the largest difference regarding data-rich assessments is its capability to deal with relatively short time series. This is aligned with the concept of data-moderate stocks proposed by Newman et al. (2015) and Geromont and Butterworth (2015), who indicated that the status of these stocks should be based on some dynamic feedback from an index of abundance. In this regard, the stock indicators derived from data-moderate stock assessments are considered reliable.

The data-limited and data-poor concepts are usually interchangeable in the literature (Geromont and Butterworth, 2015; ICES, 2012; Jardim et al., 2015a). The data-limited stock assessments include a broad spectrum of methods and consequently produce results with different accuracy levels.

The so-called “data-poor” stocks do not meet the information requirements to produce a standard quantitative stock assessment, meaning that the status involve large uncertainty compared to traditional reference points (Aylesworth et al., 2018; Dowling et al., 2016). Newman et al. (2015) add that data-poor stocks lack any feedback about current or historical  $F$ , due to their being based on static assumptions (i.e. steady state). Stock assessment methods capable to provide stock status from single-year length data and life-history parameters are an important category of data-poor or data-limited methods (Hordyk et al., 2015; Kokkalis et al., 2017, 2015; Rudd and Thorson, 2017). Additionally, the data-poor category may host those stocks where data may never meet

the requirements to conduct quantitative assessments, but a qualitative assessment may be possible (Smith et al., 2009). However, those stocks or fisheries for which advice cannot be provided through stock assessments (“data-less” stocks) are outside the scope of this thesis (Kleisner et al., 2012; Osio et al., 2015).

### 1.4. Uncertainty in the stock assessment outputs

Availability of data is not a sufficient criterion to discriminate between categories of stock assessment. This was noted by the International Council for the Exploration of the Sea for European stocks (ICES, 2012) that splits the stock assessment in six categories according to availability and quality of data. Category 1 only includes those stocks where full analytical assessments, including production models, may be performed. Stock assessments performed by quantitative methods but for different reasons only consider trends in fishing mortality, recruitment and biomass were included in category 2. Implicitly, the distinction between category 1 and 2 suggests that uncertainty in the stock assessment outputs arising from data quality and model performance deserve special attention. Categories 3, 4 and 5 are dedicated to stocks for which only survey indices, time-series of catch or landings are available, respectively. These three categories are considering only availability of data, mainly assuming proper data quality. The category 6 defined by ICES (2012) goes away from the availability and quality of data but focuses on stocks where landings are negligible compared to discards.

The Blackspot seabream (*Pagellus bogaraveo*) in Subarea 10 (Azores grounds) was advised by ICES using a data-limited approach (category 3) because the reference points are undefined. Accordingly, the Azorean bottom longline survey was used to produce an index multiplied by the recent advised catches that allows comparison between two periods (2017-2018 and 2014-2016) (ICES, 2019). The Grey gurnard (*Eutrigla gurnardus*) in Subarea 4 and divisions 7.d and 3.a (North Sea, eastern English Channel, Skagerrak and Kattegat) was also advised by ICES as data-limited stock (ICES, 2018). In case of this species  $F_{msy}$  proxy was achieved using the expected mean length of catch above  $L_c$  when  $F = M$ , assuming  $M/K = 1.5$ .

The Pacific Fishery Management Council (PFMC) that manages federal fisheries off the United States west coast classifies groundfish assessment analyses into three

categories. The type of assessment and uncertainty related to harvest specification were used to determine these categories (Wetzel and Punt, 2015). Category 1 includes stocks that may be assessed by conventional catch-at-age models, assuming to produce the lowest uncertainty for the stock assessment outputs. Stocks that belong to category 2 require to incorporate methods capable of dealing with limited data, meaning that stock status holds a moderate level of uncertainty. Category 3 includes stocks, which have limited data (e.g. only time series of catch) and abundance indices are not available. Therefore, the status of stocks in this category is surrounded by the highest uncertainty. Stocks subject to any of the three kinds of status uncertainty are managed from the acceptable biological catch (ABC), expected to be equal or lower than the overfishing limit (OFL).

Data-poor assessment methods (category 3) were adopted by the Council in 2011 (Berkson et al., 2011). Here was specified that the assessment methods to inform OFL are 1) Depletion Corrected Average Catch (DCAC), Depletion Based Stock Reduction Analysis (DB-SRA), and Simple Stock Synthesis (SSS). Nevertheless, the application of these methods is recommended for long lived species (mainly for DCAC) assuming a catch history properly known (including discards), covering the earliest fishery harvesting (mainly for DB-SRA). The terms of reference for the groundfish and coastal pelagic Species stock assessment review (Process for 2019-2020) holds the use of DCAC, DB-SRA and SSS to advice the OFL while none steady-state based method was considered (PFMC, 2019). The online repository from the PFMC do not include the reports of stock assessments of data-poor species, focusing in data-moderate and data-rich stocks (PFMC, 2017).

Australian fisheries are assessed and managed using the “tier” system, implying that as higher is the tier level, the stock is categorized as more data-limited, greater its status uncertainty and larger the risk of overfishing (Dowling et al., 2016). Tiers were originally based on four levels that describe the ability of data to produce reliable assessments, helping to define the assessment method and associated harvest control rule (Smith et al., 2008). Dowling et al. (2016) extended the tier levels from four to eight, using availability and quality criteria (i.e. stock assessment reliability). Tier 0 provides robust estimates of  $F$  and  $B$  by conventional methods, while stocks in tier 1 have only

fishery-dependent data but assessments are considered also robust. Tier 2 produces estimates of  $F$  and  $B$  but outputs are considered to be less robust. Tier 3 leads to estimates of  $F$  based on size and/or age data, being the last level that may provide some estimate of absolute  $F$  value. Tiers 4 and 5 may produce empirical estimates of  $B$  from catch-per-unit-effort (CPUE) data or fishery-independent surveys and of  $F$  deducted from relative effort. Finally, tiers 6 and 7 are not focused in producing estimates of  $F$  and  $B$ , taking management decision based on fishery-dependent triggers from species-specific or for groups of species, respectively.

The tier system, however, actually uses tier levels 1, 3 and 4 to determine a recommended biological catch (RBC) because the tier level 2, which was applied to a less robust quantitative assessments, is no longer being used (Day, 2019). This also implies that the tier system proposed by Dowling et al. (2016) is not formally enforced yet. The tier 4, as more data poor condition, is based totally on catch and CPUE. CPUE referent points are considered as proxies for the estimated Biomass limit (Blim) and Biomass target ( $B_{TARG}$ ), assuming that CPUE is proportional to stock abundance (a strong assumption, see Pauly et al., 2013). This later implies that CPUE to the start of the fishery is known, unbiased observed, and it is a proxy to the unexploited biomass ( $B_0$ ). Given that most of the stocks were not observed since the start of the fishery, reasonable estimate of the level of depletion of the stock may also be assumed into the analysis (Day, 2019). Blue-eye Trevalla (*Hyperoglyphe Antarctica*), Blue Warehou (*Seriolella brama*) and Deepwater Shark Basket – East, are three of the thirteen stocks assessed according to the Tier level 4 (AFMA, 2018). However, data-poor stocks represent only 38% of the stock assessed in the Southern and Eastern Scalefish and Shark Fishery by the Australian Fisheries Management Authority.

According to the stock assessment categories used to define the status and status uncertainty of European, North American (West coast) and Australian stocks, it is noticeable that they are not entirely comparable. This probably occurs because the nature of the fishery data (e.g. quality of time-series of catch) and consequently fishery management (e.g. enforcement of ACL) among these world-developed areas are different. However, fisheries in these areas share the existence of more or less long time-series of catches and abundance indices, promoting that the characteristics,

availability and quality of this kind of data become a key aspect to categorize the stock status uncertainty.

Some resources managed by intergovernmental institutions advising management of transboundary species also hold fragmented knowledge on the biology and fisheries. For instance, the International Commission for the Conservation of Atlantic Tunas (ICCAT) includes 13 tuna and tuna-like species in the “small tunas” category (ICCAT, 2018). This category groups species where a quantitative stock assessment cannot be performed. Currently, these species are advised from fishery indicators that use available length distributions and knowledge of life-history traits. Asymptotic length ( $L_{\infty}$ ), length at 50% mature ( $L_{50}$ ) and estimates of the size at which a cohort reaches its maximum biomass ( $L_{opt}$ ) and its proxy ( $2/3 \sim L_{\infty}$ ) are used to provide preliminary information on the status of the fishery. Nevertheless, ICCAT realize that the available information should be improved and used to run data-poor models able to provide a better advice (ICCAT, 2018).

It is no coincidence that historical times series of catches (i.e. total catch, landings and discards, catch-at-age, CPUE, abundance index, among others) are the basis of the stock assessment categorization since this kind of information is often available in the developed world (Pauly and Zeller, 2017a, 2017b) and the related-uncertainty may be explored (Sagarese et al., 2018). However, most of the stocks in Central America, South America, Africa and Asia are unassessed, coinciding with the bulk of the world small-scale fisheries (Costello et al., 2012; Pauly and Zeller, 2017a). From the stock assessment point of view, in the developing world hardly exists suitable time-series of catches, while available useful data to provide information on stock status usually covers one or sparse years. Thus, the data-limited stock assessment methods used to define the stock status in developed countries, although may greatly contribute to improving the knowledge of the status of small-scale fisheries, does not fit the whole context of these fisheries.

The classification of stock assessments used by ICES (ICES, 2012), the Pacific Fishery Management Council (Wetzel and Punt, 2015) and Australian Government (Dowling et al., 2016) highlight that quality of catch history and abundance indices are the main criteria used to define the uncertainty in the stock assessment outputs in

developed countries. Implicitly, it is assumed that life-history traits are known (including their variability). In order to include the availability and quality of life-history parameters as an additional criterion to define the status and uncertainty in stock assessment outputs of small-scale fisheries, in this thesis I proposed a particular categorization of the concepts of data-rich, data-moderate, data-limited and data-poor fisheries, considering previous classifications. Thus, four categories according to availability and quality of data and their impact on the stock assessment outputs are considered in this work:

Category 1. Stocks assessed by catch-at-age methods fed with reliable fishery-dependent and independent data and known life-history parameters (Table 2). Uncertainty in estimates of fishing mortality, stock size and stock status is low or low-moderate, depending on extension of time series of catch and surveys (Fig. 3). These stocks can be considered data-rich.

Table 2. Categorization proposed in this thesis regarding availability and quality of input data of the stocks that are subjected to stock assessment. Available data may hold proper (green) or deficient (yellow) data quality.

		Data-rich	Data-moderate	Data-limited	Data-limited	Data-poor
Availability (text) and quality (color) of data	Catch at age	Available	Unavailable	Available	Available	Available
	Catch history	Available	Available	Available	Available	Available
	Abundance index	Available	Available	Unavailable	Unavailable	Unavailable
	Life-history parameters	Available	Available	NA	Available	Unavailable
Uncertainty	Low	Category 1				
	Low-moderate	Category 1	Category 2			
	Moderate		Category 2			
	Moderate-high			Category 3	Category 3	
	High					Category 4

Category 2. Catch-at-age or catch-at-size data are not available but there is reliable information on catch history and abundance indices (Table 2). Life-history traits, including carrying capacity and stock depletion over time, are known or may be inferred. Methods based on surplus production and depletion models are suitable and uncertainty in estimates of fishing mortality, stock size and stock status, are moderate or moderate-low, depending on quality of input data. These stocks can be considered data-moderate.

Category 3. Stocks that may hold appropriate quality of some fishery data (e.g. catches or life-history parameters) but not all data is suitable to perform conventional catch-at-age or surplus production methods. For instance, there is reliable catch history but abundance indices are not available, or life-history parameters are known but catch-at-size data is either not available or available over sparse years (Table 2, Fig. 3). These stocks may be assessed by catch-based methods or require to assume steady-state. Definition of the stock status, fishing mortality and stock size carry moderate-high uncertainty. A stock that does not have catch history information but its life-history parameters have been properly estimated falls into this category. These stocks can be considered data-limited.

Category 4. There is no reliable information on catch history nor life-history parameters (Table 2, Fig. 3). Stock assessment methods based on length data may be performed but estimates of one of several life-history parameters may be biased or should be “borrowed” from other species (see below “Robin-Hood” approach). Estimates of fishing mortality (usually in relative terms), stock size (also in relative terms) and stock status involve high uncertainty. These stocks can be considered data-poor.

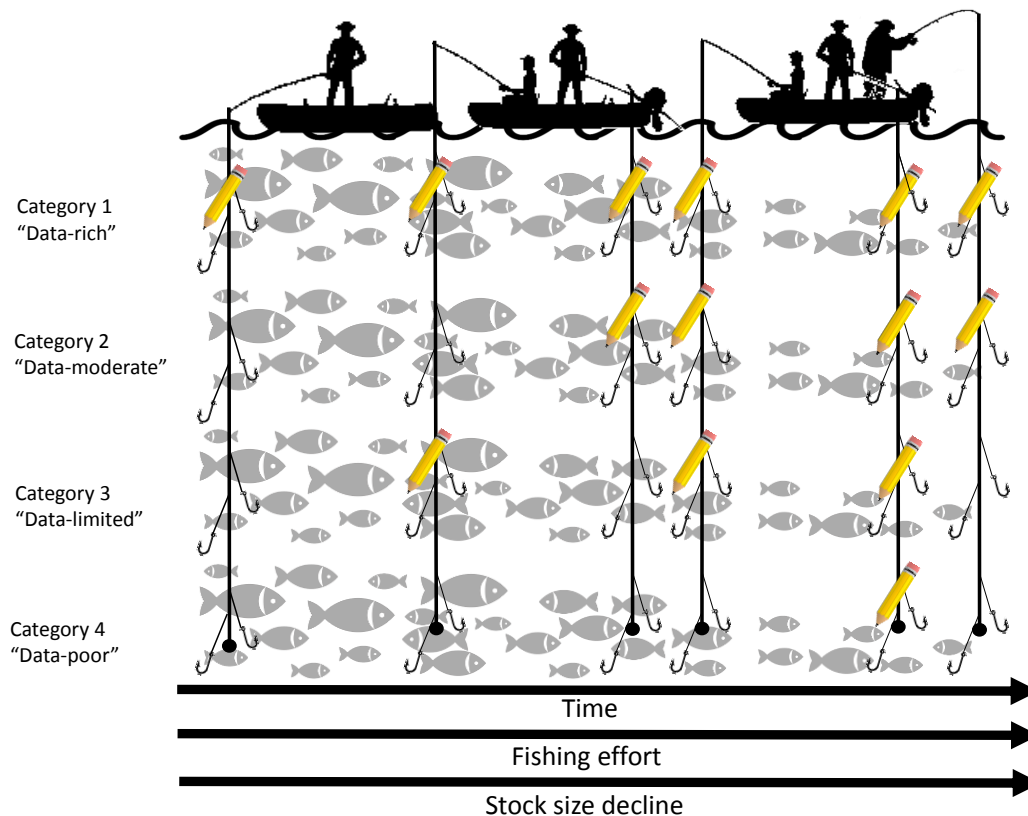


Fig. 3. Illustration of a stock that has declined over time (less fish and smaller sizes in recent periods) while fishing effort have increased (showed as an increase in number of fishers by boat). The stock is used to illustrate the stock assessment categories according to the availability and quality of data. Pencil indicates the data availability. Figure elaborated by John Ramírez.

### 1.5. Bias in data and information derived from small-scale fisheries

The Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries (FAO, 2015) signed by all countries that belong to the Food and Agriculture Organization of the United Nations avoided to define the small-scale fisheries (SSF) concept. This occurred because it was recognized that the criteria used (e.g. size of boats, method of production, level of organization, power to remove biomass, among others) to attempt to achieve a uniform definition fit unequally the world-wide small-scale fisheries. However these guidelines highlight that SSF contribute about two-third of global fish catches aimed for direct human consumption and employed at least 90% of the world's fishers and fish workers. Although the importance of SSF is beyond doubt (e.g. Fig. 4), assessment and science-based management of these fisheries is still marginal (Costello et al., 2012; Pauly, 2006b, 2006a; Teh and Pauly, 2018).











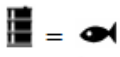



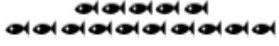

FISHERY	LARGE SCALE 	SMALL SCALE 
BENEFITS		
Number of fishers employed	 about 1/2 million	 over 12 millions
Annual catch for human consumption	 about 30 million t	 same: about 30 million t
Capital cost of each job on fishing vessels	\$\$\$\$\$\$\$\$\$\$\$\$ 30,000 - 300,000	\$ 300 - 3,000
Annual catch reduced to meals and oils	 20 - 30 million t	 Almost none
Annual fuel consumption	 About 37 million t	 About 5 million t
Catch per tonne of fuel consumed	 1 - 2 t	 4-8 t
Fishers employed for each \$1 million invested in vessels	 5 - 30	 500 - 4,000
Fish and other sealife discarded at sea	 8-20 million t	 Very little

Fig. 4. Schematic illustration taken from Pauly (2006a) comparing large and small-scale fisheries.

The small-scale fisheries usually do not have reliable information of catch history and fishery-independent abundance indices are not available. Therefore, stock assessment based on catch-at-age statistical models (category 1: data-rich stocks) and on surplus production models (category 2: data-moderate stocks) cannot be performed. From the stock assessment arena, the small-scale fisheries might be defined as those that when they may be assessed, only methods based on landings (as proxy of catch) or that assume steady-state (categories 3: data-limited and 4: data-poor) are suitable. The uncertainty in estimates of fishing mortality, stock size (when possible) and stock status of SSF is high, because of stock assessment is surrounded by strong assumptions (Hordyk et al., 2015; Prince et al., 2015c; Rätz et al., 2010) and model inputs may be biased. Salas et al. (2007) described some common characteristics to SSF that may be used as basis to help to understand the bias that the small-scale fishery-dependent data underlies (also see Table 3):

- Multi-fleet fisheries. Several kinds of boats according to size and power among other features, as well as many different gears are used by fishers to remove biomass from the sea. This implies that resources are affected by different gear selectivity (Garcia and Ramírez, 2016; Huse et al., 1999), which many times are not well recorded by the monitoring program. Therefore, length-based data used to estimate or derive life-history parameters may be biased (Gwinn et al., 2010).
- Multi-species fisheries. Particularly, in the context of SSF, catching several species could mean that one species may acquire several common names (synonyms) or many different species may be called by the same common name (homonyms) along landing points. Thus, a consistent catch history by species may be hindered by misleading reporting of catches (Freire and Pauly, 2005).
- Low levels of capital investment. Both fishers and others stakeholders (including government) dedicate low investment, to make sure a stable, comparable and continue monitoring program, either official or community-based. Consequently, catch history, CPUE and length-based data (used to estimate life-history parameters, among others) often are sparse and not entirely comparable among years.
- A wide range of landing sites. Small-scale fisheries usually land in rural areas (Salas et al., 2007), meaning that an unquantified but important portion of total landings may be underestimated when monitoring is only focused on urban landing points. Additionally, fishing effort may change between landing points and years.
- Seasonal use of fishing resources. Fishers may combine fishing and other activities (e.g. agriculture, tourism, among others) along year. Depending on benefits derived from each activity, social commitments and environmental conditions, fishing effort may vary among seasons and years. Therefore, catch history may not be suitable to be used as a proxy of stock abundance (Pauly et al., 2013). Additionally, incomplete information to estimate some life-history traits may occur because

fisheries may not provide full information on the biology of species along a year (i.e. size of maturity and gonadosomatic index).

- Source of food (protein) and jobs. Historical landings may not be assumed as proxy of historical catches since target and discarded species and those used to local consumption could have changed over time (García et al., 2007). Additionally, the portion of catch used to direct consumption usually is not reported in the fishery statistics (García, 2010).
- Attraction of migrant people to the coast in the search for income. People who have recently started to fish may prefer to use gears holding higher removal power regarding traditional gears. Consequently, fishing rates and selectivity between early and late years are not entirely comparable.
- Limited power of fishers to influence the fish market. Landings of species harvested by small-scale fisheries are often grouped in general market-price categories, making it difficult to know the particular species that are caught (Gil et al., 2018).
- Lack of social mechanisms for health and employment provision to the fishers. Subsidies may act as a fishing effort factor among years. Given that subsidies are frequently linked to political issues, the historical fishing effort perceived through statistics may be affected by this reason.

Table 3. Common biases in the information used by the stock assessment models when it is derived from small-scale fisheries.

Small-scale fishery trait	Usual condition	Definition of length structure	Length-based estimates of life-history parameters	Fishery selectivity assumptions	Knowledge on catch history	calculation of CPUE
Multi-fleet fisheries	Partial monitoring					
Multi-species fisheries	Biased species definition					
Low levels of capital investment	Partial monitoring					
Wide range of landing sites	Partial monitoring					
Seasonal use of fishing resources	Incomplete year data					
Source of food (protein) and jobs	Incomparable over time					
Attraction of migrant people to the coast in the search for income	Incomparable over time					
Limited power of fishers to influence the fish market	Biased species definition					
Lack of social mechanisms for health and employment provision to the fishers	Incomparable over time					

### 1.6. “Robin Hood” approach

In Australia, the Southern and Eastern Scalefish and Shark Fishery (SESSF) catch over 300 species, but formal assessments were only available for ten species in the early century (Punt et al., 2005). These authors indicated that the monitoring agency had insufficient technical or financial resources to assess all fishery species, even to assess those 25 species that comprised around 90% of landed catch. Therefore, prior probability distributions of virgin biomass,  $M$  and the steepness of the stock-recruitment relationship from well-assessed stocks was thought as useful basis to set the stock assessment models of data-poor stocks (Punt et al., 2005). In this context, the data-poor concept (redefined by us as category 3) was allocated to stocks holding not too long but reliable time series of catch. Thus, a Bayesian framework within which the assessments results of data-rich species were used to allow assessments for data-poor fisheries was implemented (Punt et al., 2005; Smith et al., 2009). This approach was termed the “Robin Hood” approach in Australia, where data-rich information on prior probability distributions of key stock parameters are “stolen” to “give” it to data-poor species (Scandol et al., 2009; Smith et al., 2009).

Essentially, whether multiple stocks are harvested by the same fleet, the trend in fishing mortality of these stocks could be expected to be similar but not identical (Punt et al., 2011). Therefore, the “Robin Hood” approach considers stock-specific parameters in all stock assessment models, but penalties are used to include variation for these parameters among stocks. The penalties were included in the model to 1) recruitment deviations, 2) length-at-50% of selectivity and 3) relative trend in exploitation rate (Punt et al., 2011). These authors found that stock assessment outputs of data-poor stocks were markedly different when penalties were used. Thus, constraining assessments of data-poor fisheries reduced the change of stock assessments by following the noise that small datasets underlie. Finally, advice on estimates of spawning biomass and exploitation rate may be provided to data-poor fisheries (Carruthers et al., 2014).

The “Robin Hood” approach has been implemented in order to assess data-limited stocks (according with the redefinition of the data-limited concept in the current thesis) that are jointly caught with data-rich stocks. However, most of the stocks caught by small-scale fisheries lack information of catch history. Therefore, the “Robin Hood” approach cannot be implemented in the way that it was initially drawn (Punt et al., 2005, 2011; Smith et al., 2009). To assess small-scale fisheries, Prince et al. (2015a) performed a meta-analysis to find robust estimates of growth, natural mortality, size of maturity and length-weight relationship. Thus, these authors “borrowed” the best available information on life-history parameters of related species to set the length-based spawning potential ratio model (LB-SPR). In a wide context, the work of Prince et al. (2015a) and other authors to estimate, for instance, natural mortality (Kenchington, 2014; Then et al., 2015) are an extension of the “Robin Hood” approach. However, the condition of using “priors” of the data-rich stocks that were jointly fished with the data-poor stock is not fulfilled.

### 1.7. Objectives and structure of the thesis

This thesis aims at addressing the reliability of the data-poor stock assessment methods to provide advice on the status of small-scale fisheries lacking catch history. In order to achieve this aim, the four studies in chapter 2 are dedicated to explore the stock assessment results under different availability and quality of data. In this thesis the stock indicators of the European hake (*Merluccius merluccius*) obtained by a variant of the

virtual population, the eXtended Survivor Analysis (XSA) that expresses the results in terms of survivors, avoiding the error-prone projection through the final year (Shepherd and Pope, 2002), are compared to those estimated by pseudo-cohort VPA (data-limited stock assessment) (Chapter 2.1). This species is also assessed by LB-SPR (data-poor stock assessment) (Chapter 2.2). Posteriorly, these two stock assessment methods are used to determine the status of three unassessed species under different availability and quality of data. These species are targets of data-poor small-scale fisheries in Colombia (*Lutjanus synagris* and *Haemulon plumierii*, chapter 2.3) and Peru (*Anisotremus scapularis*, chapter 2.4). The four chapters are used to discuss the findings in the light of the main objective of the thesis, but also to provide insights into the current status of assessed species according to their specific fishery context.

The general objective of this thesis and specific objectives of the four above-mentioned chapters are the following:

### General Objective

To determine the reliability of the data-poor stock assessment methods to provide advice on the status of small-scale fisheries lacking knowledge of catch history.

### Specific objectives

1. To determine the ability of pseudo-cohort VPA to provide reliable indicators of harvesting and stock size by treating the Northwest Mediterranean European hake as data-limited stock.
2. To assess the exploitation rate of hake in the Southwestern Mediterranean based on a size frequency analysis and basic biological information under different growth hypothesis using the LB-SPR method.
3. To improve stock assessment and management advice for data-poor small-scale fisheries through participatory monitoring, using the Wayuu fisheries in the Northeastern Colombian Caribbean Coast as case study.

4. To address the impact of the uncertainty in estimates of life-history parameters on the spawning potential ratio outputs of the data-poor species Peruvian grunt in the central coast of Peru.

Chapter 2.1 compares the stock indicators of the European hake in Northwest Mediterranean derived from pseudo-cohort (Leonart and Salat, 1997) and XSA (Shepherd, 1999). This chapter explores how accurate are the pseudo-cohort VPA results when stock assessment is performed under steady-state conditions assuming unbiased data. This study investigates the uncertainty derived from setting terminal fishing mortality ( $F_{ter}$ ), which is a “guessed” input value in the pseudo-cohort VPA. The importance of exploring this value relies on the fact that it may promote large changes in stock size, largely problematic when the pseudo-cohort analysis is performed to provide temporal trends of sparse years. Using single-year data, this method may be capable to provide the same stock indicators than those provided by formal catch-at-age methods (e.g.  $F$ , stock number,  $SSB$ ). Therefore, stock assessment of species other than the target of multi-species fisheries (e.g. small-scale or trawl fisheries) could provide additional support to propose proper management measures. This study provides insights to understand the bias that stock assessment under data-limited assumptions may face when the pseudo-cohort VPA is used.

The chapter 2.2 assesses the status of the European hake (Box 2) using only size frequencies and basic biological parameters in the Southwest Mediterranean. Here is used the LB-SPR model (Hordyk et al., 2015; Prince et al., 2015a), assuming steady-state conditions, again. This analysis also assumes that catch history is not available, thus stock assessment is performed for each year in isolation. The size structure of the harvested stock was defined by more than 2000 samples every year. Additionally, the life-history parameters required as model inputs were considered accurate and known. This quality of data and information are not usually available in small-scale fisheries, offering a good opportunity to comprehend the impact of data quality when data-poor stock assessment methods are used. Finally, the data-limited stock assessment results arising from the  $M/k$  ratio instead of each individual parameter are investigated when the fast- and slow-growing hypothesis are considered. This is particularly relevant to the

European hake stocks in the Mediterranean Sea since there are several studies that support both hypotheses.

Box 2 Bottom trawl fishery of hake in West Mediterranean (Chapter 2.1 and 2.2)

**Study area:** The bottom trawl fleet harvests target and bycatch species along Mediterranean waters (Osio et al., 2015). Particularly, the chapter 2.1 and 2.2 of this thesis have used data from the southwestern (GSA1) and northwestern (GSA6) Spanish Mediterranean Sea (Figure a).

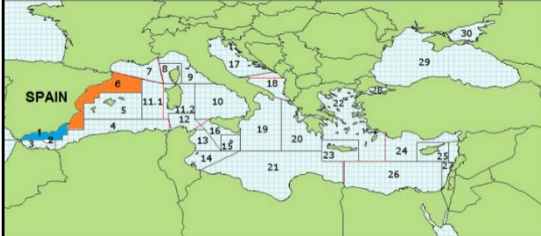


Figure a. Geographical subareas (GSA) in the Mediterranean and Black Sea. South (GSA1) and north of Spain (GSA6) are shown in blue and orange, respectively.

**Fishery:** Hake has been one of the most fished demersal species in Mediterranean waters (Figure b) and is one of the targets of the Bottom trawl fishery (Figure c) (Bas et al., 1955; Cardinale and Scarcella, 2017; Colloca et al., 2013; STECF, 2015). This species produced annual average catches from 2003 to 2014 of 462 t (Stand. Dev. 130 t) and 3677 t (Stand. Dev. 779 t) in GSA1 and GSA6, respectively (STECF, 2015).



Figure b. *Merluccius merluccius* (ES: “merluza”, EN: “European hake”) (European Commission, 2018).



Figure c. Example of Bottom trawler used for harvesting European hake in the Spanish Mediterranean Sea (Eldiariocv.es, 2016).

**Life-history parameters:** The von Bertalanffy growth parameters ( $vBGP$ ) of the European hake have been estimated using length-frequency, tag-recapture and otolith-reading technics (Aldebert et al., 1993; Aldebert and Recasens, 1996; Martínez-Baños et al., 2018; Mellon-Duval et al., 2010; Piñeiro et al., 2007). The  $vBGP$  have been used to estimated natural mortality by indirect estimators (Abella et al., 1997; Aldebert et al., 1993; Aldebert and Recasens, 1996). Maturity at size is often updated and used in the official stock assessments (STECF, 2015).

**Stock assessment:** The bottom trawl fishery of hake in GSA1 and GSA6 is assessed from the Data call framework (DCF) of the European Union by catch-at-age stock assessment methods since 2008 (STECF, 2015).

Fishery-independent (surveys) and fishery-dependent data (fishery) is available since 2003. The spawning stock biomass of both stocks has tended to decrease (Figure d1) (STECF, 2015). The most fished ages ( $Fbar$ ) indicated overfishing (Figure d2).

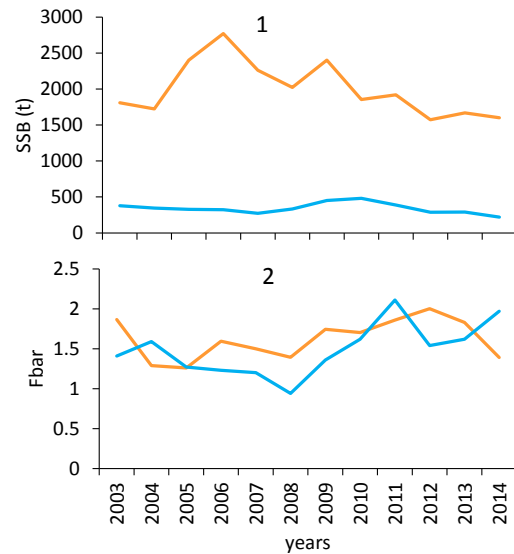


Figure d. Stock assessment results of the European hake in GSA1 (blue) and GSA6 (orange).1) spawning stock biomass and 2) the most fished ages ( $Fbar_{0-2}$ ) and ( $Fbar_{1-3}$ ) in GSA1 and GSA6, respectively (STECF, 2015).

Most recent stock assessment of hake was carried out combining GSA 01, 05, 06 and 07 stocks (STECF, 2018).

Chapter 2.3 investigates the impact of missing and misunderstood data on single-species stock assessments used to provide management advice of small-scale fisheries. Considering the biases underlying the monitoring of small-scale fisheries and how these biases are propagated into the stock assessment model inputs, the highest-valued fish (lane snapper) and the largest-landed species (white grunt) of the Wayuu fishery in the Colombian Caribbean were assessed (Box 1). The status of these two data-poor species was defined using pseudo-cohort VPA. Then, several management scenarios based on fish size and deployed fleet were simulated. To date, the stock assessments performed by the pseudo-cohort VPA have assumed that input data is unbiased, while bias in the model inputs has not been explicitly investigated. Given that stock assessment of data-poor small-scale fisheries usually is hampered by the availability and quality of data, this chapter focuses on these issues. The role of participatory monitoring to improve the stock assessment of small-scale fisheries is carefully considered. Additionally, this study focuses on exploring how advice could be provided from the single-species stock assessment arena to a small-scale fishery with more than one fishery target and employing several gears.

Chapter 2.4 deals with the status uncertainty derived from data-poor stock assessment when life-history parameters are unknown and only a small set of length frequencies of the fishery target is available. This data availability supposes the more extreme situation in stock assessment but is common in small-scale fisheries. Under these circumstances, the uncertainty in the stock status estimated by methods quite sensitive to misspecification of life-history parameters, such as the LB-SPR model, is very high (Hordyk et al., 2015). Therefore, several methods to estimate von Bertalanffy growth parameters and natural mortality are employed. These parameters are used to set the inputs model in order to comprehend the impact of inputs uncertainty on the status of a data-poor stock. The small-scale fishery that is investigated targets the Peruvian grunt in the central coast of Peru (Box 3). Given that this fishery is mainly carried out by beach fishers and that government monitoring programs collect fishery data in the main ports only, participatory monitoring was required in order to obtain fishery data of this species. From these results, the scope of stock assessment on data-poor species is discussed.

### Box 3. Beach fishery of Peruvian grunt in Peru (Chapter 2.4)

**Study area:** The anchovy fishery in Peru is one of the most important world's fisheries (Chavez et al., 2003). Nevertheless, other fisheries are key in local economy of Peruvian fishers (Christensen et al., 2014). This is the case of the Peruvian grunt (*Anisotremus scapularis*, Figure a) that is mainly harvested by artisanal fishers (around 5300 fishers in Peru), ranking the second place in the most fished species using beach gears (INEI, 2012).

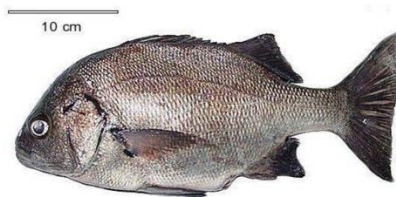


Figure a. *Anisotremus scapularis* (ES: "Chita", EN: "Peruvian grunt") (INFOPE, 2015).

The Peruvian grunt inhabits mostly breaking zones (Angel and Ojeda 2001; Perez-Matus et al. 2007). The fishing monitoring carried out by the Peruvian Government takes place in fishing ports (Guevara-Carrasco and Bertrand, 2017). Therefore, landings of the Peruvian grunt may be seriously underestimated. The chapter 2.4 studied the Peruvian grunt landed by the beach fishers in the central coast of Peru (Departments of Lima and Ica) (Figure b).

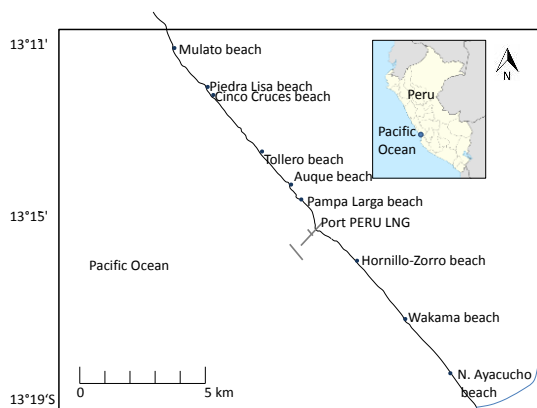


Figure b. Study area.

**Fishery:** The Peruvian grunt in the central coast of Peru is harvested using 12 artisanal gears based on hooks and gillnets (Figures c and d).



Figure c. Fishery of Peruvian grunt using hooks (picture: John Ramírez)



Figure d. Fishery of Peruvian grunt using gillnets (picture: John Ramírez)

Landings reported in the fishing ports have decreased by half from near 200 tons (2000) to almost 100 tons (2012) (Guevara-Carrasco and Bertrand, 2017). IMARPE (2018) highlights the local importance of this species but recognize that the fishery and biology information is still incipient. Additionally, IMARPE suggested an unquantified underestimation of landings from beach fisheries.

**Life-history parameters:** The von Bertalanffy were recently estimated. It is assumed a single growth curve along the Peruvian coast (latitudinal gradient encompassing from 4° to 18° south) (IMARPE, 2018), while an outstanding effect of water temperature on the growth of this species was already evidenced (Dionicio-acedo et al., 2018). Size of maturity of Peruvian grunt was estimated in 21 cm (L50) IMARPE, 2018.

**Stock assessment:** Using several data-poor methods based on length distribution data and life history parameters the stock status was defined as unsustainably exploited (IMARPE, 2018). Data uncertainty was not considered.

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## 2. RESULTS



2.1. May steady-state Virtual Population Analysis (VPA) provide reliable estimates of stock indicators compared to conventional VPA? : European hake treated as data-limited stock



**May steady-state Virtual Population Analysis (VPA) provide reliable estimates of stock indicators compared to conventional VPA? : European hake treated as data-limited stock**

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**Abstract**

We examine whether the pseudo-cohort VPA that works with single-year data may provide comparable results to those derived from the eXtended Survivor Analysis, a variable of VPA that overcomes issues related to *ad hoc* tuning procedures. We treated the European hake in the North-western Mediterranean as a data-limited species. A slight variation of young classes among years was translated into larger uncertainty on stock size, meaning that terminal fishing mortality ( $F_{ter}$ ) had a notable impact on the data-limited VPA outputs. Remarkable agreement on the trend of stock indicators as well as reference points was found between two methods (relative differences < 0.2, 2005-2014). Sparse data available since 1982 suggested that fishing mortality has not decreased over four decades while the stock size has depleted by half. Our findings suggest that the pseudo-cohort VPA may be a useful tool to assess the trend on the stock status of other than target species of multi-species fisheries.

**Keywords**

Data-limited, stock assessment, stock indicators, *Merluccius merluccius*, Mediterranean fisheries, terminal fishing mortality.

**Introduction**

Conventional fisheries stock assessment is based on mathematical models of population dynamics, among which Virtual Population Analysis (VPA) is the standard in regions of the world with a long tradition of fisheries assessment and management

(Gulland 1965; Hilborn and Walters 1992; Lassen and Medley 2001) (but see also Carruthers et al., 2014). Classical VPA requires, at minimum, input data sets with catch-at-age in number by fleet and biological information on growth (weight-at-age in the catches and the population), and natural mortality and maturity ogives at age (e.g. Hilborn and Walters, 1992; Shepherd and Pope, 2002 for standard presentations of the method). The catchability  $Q$  (multiplicative calibration constant) is unknown to catch data, while that the terminal population (additive constant) is unknown to abundance indices (Shepherd and Pope 2002). However, both constants may be determined by putting the two sorts of information together, leading subsequently to estimate of the absolute population size. This procedure is commonly called as “tuning” of VPA using survey data. Naturally, catch data and biological parameters should be representative of all fish removed from the stock, including discards, over the longest time series possible to produce reliable results. Data series that span less than at least one generation are usually considered unsuitable to produce proper fisheries advice.

Due to the needs of providing fisheries advice in less than ideal situations (ICES 2015), VPA based on pseudo-cohorts has sometimes been used as alternative when the only available catch at age data spans one or a few consecutive years (Jones 1984). Early stock assessments in Mediterranean fisheries in the 1990s relied heavily on VPA based on pseudo-cohorts, using the VIT software (Virtual population analysis, Interaction and Transition) developed by Leonart and Salat 1992; 1997. Some early applications are found in, among others: Aldebert *et al.* (1993), Demestre and Leonart (1993), Oliver (1993), Pertierra and Leonart (1996) and Sardà and Leonart (1993); as well as, more recently, in stock assessment reports of the SubCommittee for Stock Assessment of the GFCM (General Fisheries Commission for the Mediterranean) (GFCM 2012). This situation derived from the paucity of data sets with sufficiently long time series of commercial landings samplings and/or fisheries independent surveys data to apply standard formulations of VPA, as used routinely in stock assessment in other geographical areas. With the implementation of standardized Mediterranean sampling programmes (DCR: EU reg. 1543/2000, and then Data Call Framework (DCF): EU reg. 665/2008) the assessment of the Mediterranean fishery targets moved, since the last

decade approximately, to a variant of VPA (eXtended Survivors Analysis, XSA: Shepherd, 1999), in line with the standard assessment practices of the ICES community.

XSA focuses on determining the surviving population for each cohort, and through using VPA estimates the past population abundance (Shepherd and Pope 2002). Thus, the error-prone projection through the final year derived from the analysis of the catchability by *Ad hoc* methods for tuning VPA is avoided. Although XSA continues to assume that catch is unbiased during the calculation phase, as occur in the earlier VPA methods, accepts that survivors deviate from the population estimates. Thus, the XSA method is close to the so-called integrated methods (Methot Jr and Wetzel 2013; STECF 2017), while it controls data demand.

The data-limited VPA (here VIT), which is based on pseudo-cohorts assumes that the abundance of the age groups of a given year correspond to the age groups of a single cohort along its life span. Therefore, fishing mortality, natural mortality and recruitment among years are assumed constant. These assumptions were called by Sparre et al. (1989) as “constant parameter system”. These assumptions have serious methodological limitations (Rätz et al. 2010), but may still be a valid alternative in the analysis of “data-limited” fisheries. As conventional VPA, VIT also requires setting terminal fishing mortality ( $F_{ter}$ ) to start the back calculation of fishing mortality and stock size. Contrary to VPA that performs several sensitivity analysis in order to properly define  $F_{ter}$  (Darby and Flatman 1994), VIT considers  $F_{ter}$  as a “guessed” input (Leonart and Salat 1997).  $F_{ter}$  may have strong impact on the resulting fishing mortality and stock size estimates, especially if through its estimation the convergence property of VPA is affected or destroyed (Shepherd and Pope 2002). Previous studies carried out with VIT prioritized finding stable trends in the older part of the  $F$  vector as criterion to set  $F_{ter}$  (Demestre and Leonart 1993; Aldebert and Recasens 1996). However, the impact of  $F_{ter}$  value when stock indicators are compared among years assuming a steady state has been never explored.

Rätz et al., (2010) compared the performance of VIT with XSA in a data-rich fishery (cod stock in the Skagerrak, North sea and Eastern Channel) and found that the overall increase in  $F_{bar}$  in the 30 year period 1963-1992 was well captured by VIT, although they pointed the low precision of annual estimates (differences of  $\pm 15$  to 20%).

However, the precision was much better for the first age classes, which suggests that VIT correctly estimated recruitment. Additionally, this comparison found that VIT tends to systematically underestimate *SSB*. Aldebert and Recasens (1996) reported a high congruence between VPA (ADAPT model) and VIT analysing a short time (1988-1991) series of hake landings in the Gulf of Lions.

VIT software provides, from VPA and Y/R analysis, estimates of stock size (e.g. *SSB/SSB<sub>0.1</sub>*) as well as stock status (e.g. *F<sub>0.1</sub>*) regarding fishing mortality and biomass reference points. The stock estimators and referent points provided by VIT are comparable with those found through XSA method. Thus, assessing and monitoring other species than the target species under comparable fishery indicators may help refine the management strategies enforced for fleets removing many species (e.g. multispecies fleets such as those using bottom trawl). Especially, it may be useful in areas (e.g. Mediterranean Sea) where the target species have been frequently assessed by VPA methods (e.g. XSA), but species other than target species are not assessed because do not accomplish the data requirements of these methods. VIT may also be suitable for defining the stock status of unassessed fisheries or in cases where a fisheries-independent abundant index cannot be reliably used because the species exhibits a very contagious concentration pattern (STECF 2017).

For most of the species caught by trawl and mixed fisheries adequate financial resources probably will be never allocated to obtain reliable and continuous time series of catch-at-age data and abundance indices. Additionally, the life-history of the species that hold a "secondary" role in these fisheries are often poorly estimated or unknown. Conversely, sparse length-frequency data and life-history information borrowed from taxonomically-related species could be used to infer the relative size and status of these stocks under comparable stock indicators. The aim of this study is to investigate if the pseudo-cohort VPA (steady-state assumptions) may provide reliable estimates of stock indicators compared to XSA, which is assumed to provide free-error estimates. We treated the European hake in the Northwest Mediterranean (GSA6), which holds accepted stock assessments based on XSA, as a data-limited resource to (i) examine the stock indicators trends between the pseudo-cohort (VIT) and extended survivor analysis (XSA), (ii) compare VIT and XSA outputs to determine the impact of uncertainty in  $F_{ter}$

and relative error on the annual estimates of 12 stock indicators and four reference points in a data-limited framework, (iii) rebuild the historical trend of stock indicators from sparse data since 1982 and (iv) determine the sensitivity of the pseudo-cohort VPA to misspecification of life-history parameters. Finally, we discuss the scope and limitations of using the pseudo-cohort VPA to obtain information on stock indicators in data-limited situations lacking long and continuous time series.

### **Material and Methods**

#### Input data

In order to perform the comparison between the data-rich (XSA) and data-limited (VIT) VPA, we use the same input data in the two models (Table 4). Life-history parameters of European hake were used as accepted by the Scientific, Technical and Economic Committee for Fisheries (STECF, 2015). This means that the estimates of von Bertalanffy growth parameters (*vBGP*), natural mortality at age (*M-at-age*), maturity at age and *a* and *b* constant values of length-at-age relationship were assumed as unbiased.

Landings from 2002 to 2015 were obtained from the official assessment reports (García-Rodríguez et al. 2006; STECF 2015). Catch-at-age data (2005-2014), which was derived from the DCF (STECF, 2015) was used as input of the XSA. This data corresponded to the samplings of commercial trawlers in GSA06 (Table 5). The VIT was fed with length frequency distributions data (from the DCF, STECF 2015), but the stock assessment was performed using catch-at-age numbers (as used in XSA). Catch-at-size data were transformed to catch-at-age, with the size to age conversion (“slicing”) built in VIT (Lleonart and Salat 1997).

Table 4. Input data and parameterizing of the XSA and pseudo-cohort analysis (VIT software) models to assess the European hake in Mediterranean Sea.

Inputs	Parameterization	XSA	VIT
<b>Life-history parameters</b>			
$vBGP$	For females, males and undetermined individuals combined (Mellon-Duval et al. (2010), $L_{\infty} = 110$ cm, $k = 0.178$ $y^{-1}$ , $t_0 = 0$ yr)	X	X
$M-at-age$	For ages from 0 to 5+ (1.24, 0.58, 0.45, 0.40, 0.37 and 0.35) by Prodbiom (Abella et al., 1997)	X	X
Length at weight relationship	$W = aL^b$ , where $a = 0.00677$ and $b = 3.03509$ (STECF, 2015)	X	X
Maturity at age	For ages from 0 to 5+ (0, 0.15, 0.82, 0.98, and 1.0) (STECF, 2015)	X	X
<b>Fishery-dependent data</b>			
Landings	From 2002 to 2014 (García-Rodríguez et al. 2006; STECF, 2015)	X	X
Discards	Discards were considered negligible	X	X
Fleet	Bottom trawl. Set nets and longline fleets had a marginal contribution (annual mean < 5%) (STECF, 2015). These two fleets were not considered in VIT	X	X
Catch-at-age	From STECF, 2015	X	
Catch at length	By combining catch-at-length number and landings. Catch-at-length data (STECF, 2015) were transformed to catch-at-age using the size to age conversion ("slicing") built in VIT		X
<b>Fishery-independent data</b>			
Abundance index	Tuning" of VPA	X	
<b>Model settings and sensitivity</b>			
$F_{ter}$	Used as input data for starting the back calculation from the number of individuals at the beginning of the last age.		X
$r_{age}$	Age at which catchability is considered to be independent of year class strength ( $r_{age} = 1$ ) (STECF, 2015)	X	
$q_{age}$	Age above which catchability is independent of age ( $q_{age} = 2$ ) (STECF, 2015)	X	
$shk_{age}$	Examining biases in assessment predictions of $F_{ter}$ in terms of ages ( $shk_{age} = 2$ ) (STECF, 2015)	X	
$shk_{yr}$	Examining biases in assessment predictions of $F_{ter}$ in terms of years ( $shk_{yr} = 3$ ) (STECF, 2015)	X	

In order to provide comparable exploitation indicators for GSA06 since 1982, only data from the trawl fleet were used in the stock assessment. However, this procedure does not prevent to compare the VIT and XSA outputs given that, from 2002 to 2014, the sampled individuals and recorded landings for fleets other than trawlers (set nets and set longlines) represented a mean of 0.84% ( $\sigma = 0.74\%$ ) and 9% ( $\sigma = 3\%$ ),

respectively of the annual landings (STECF 2015). Landings for sparse years from 1982 to 2001 had to be reconstructed. These samples came from samplings representing the entire catch collected on board commercial bottom trawl vessels based on the fishing ports of Vilanova and Port de la Selva (Catalonia, northern half of GSA06). The landings in 1989 from Valencia and Catalunya, which make up GSA06, were used to estimate a raising factor of 1.82 that was applied to years where only landings of Catalunya were available (Martín 1991).

Table 5. Summary of the data used for the assessment of hake in GSA06, with source (research project and publication). Landings represent reconstructed landings for GSA 6 for the period 1982-2001 and official landings, as reported to in STECF (2015) from 2002 to 2014 Total number of individuals are the number of individuals measured.

Year	Research project	Published in	Landings (t)	Total nº individuals
1982	AQUDE	(P. Martín, pers. comm.)	3612	
1988		(L. Recasens, pers. comm.)	4507.3	23 898
1989		(L. Recasens, pers. Comm.)	3995.6	9 223
1990		(L. Recasens, pers. comm.)	4164.5	16 558
1991		Martín, 1991	4951.2	24 515
1994		(F. Maynou, pers. comm)	4448.5	
1996	Discards I		3527.4	2 461
2001	Discards II		2712.5	57 853
2002	DCF	STECF, 2015	3195	55 718
2003	DCF	STECF, 2015	4176	275 462
2004	DCF	STECF, 2015	3750	84 651
2005	DCF	STECF, 2015	4035	49 119
2006	DCF	STECF, 2015	4635	73 049
2007	DCF	STECF, 2015	3391	47 031
2008	DCF	STECF, 2015	4021	68 421
2009	DCF	STECF, 2015	5082	85 022
2010	DCF	STECF, 2015	3278	28 205
2011	DCF	STECF, 2015	3254	25 608
2012	DCF	STECF, 2015	2900	24 859
2013	DCF	STECF, 2015	3256	23 241
2014	DCF	STECF, 2015	2230	17 678

### Model settings and sensitivity

According to the official stock assessment of hake in GSA06, several sensitivity tests were performed in order to find the model that properly explained the exploitation pattern (STECF 2015). Final model settings in XSA, including examination of the biases in assessment predictions of  $F_{ter}$ , were defined as indicated in Table 4. VIT does not require to use an abundance index for “tuning” the model. However, to start the reconstruction of the age-structured stock, VIT needs to set  $F_{ter}$ . Although this parameter is considered a “guessed” input in VIT,  $F_{ter}$  was carefully explored before producing the final stock assessment output. Instead of selecting the "best" value of  $F_{ter}$  in VIT, we considered 22 different values from 0.01 to 2. By using this range we ensured that a rapid convergence occurred ( $F_{ter} > 1$ ) and explored the impact of underestimation of this value on resulting stock size (low values of  $F_{ter}$ ). Full description of the XSA and VIT models may be found in Shepherd (1999) and Shepherd and Pope (2002) and Leonart and Salat (1992, 1997), respectively.

The comparison of results was discussed in terms of fishing mortality (per age and most fished ages), stock size (recruits, stock number, initial annual biomass and initial annual spawning stock biomass) and reference points ( $F_{0.1}$  and  $F/F_{0.1}$ ). In the STECF working groups  $F_{0.1}$  is taken as proxy for  $F_{msy}$ .  $F_{0.1}$  is indicated as an acceptable approximation of management targets for sustainable fisheries with high yields and low risk of collapse (Kell and Fromentin 2007). For a proper comparison of the VIT and XSA outputs, the number of individuals at age, the von Bertalanffy growth parameters, the length-weight relationship and the maturity-at-age vector were used for recalculating the biomass ( $B$ ) and spawning stock biomass ( $SSB$ ) shown in the official STECF report (STECF 2015). This was required because we realized that the  $B$  and  $SSB$  provided in the official assessments did not correspond exactly to the population number estimated by VPA in GSA06, probably due to differences in the length to age conversion algorithms.

### Reference points

Yield-per-recruit analysis ( $Y/R$ ) was used to calculate  $F_{0.1}$ . The reference points in the official stock assessment for the assessed period were used (STECF 2015). The reference points estimated by VIT were obtained every year by dividing the total weight

of the catch by the number of recruits (age 0) calculated from the pseudo-cohort VPA. In order to express  $Y/R$  as a function of the fishing effort, the VIT software introduced the multiplier factor ( $\phi$ ), reformulating the catch equation from initial weights as:

$$\frac{Y}{R}(\phi) = \frac{1}{N_0} \sum_{i=1}^n \bar{N}_0 w_0 F_0 \phi$$

The  $Y/R$  analysis provided the multiplier factor that corresponded to  $F_{0.1}$ . Current fishing mortality ( $F_{bar}$ ) was determined as the average of the most fished ages, and  $F_{0.1}$  was calculated as:

$$F_{0.1} = \frac{F_{0.1} \phi * F_{bar}}{F_{current} \phi}$$

#### Bias estimation

In order to achieve the main goal of this work, we assumed that XSA produces unbiased estimates of the stock indicators. This means that determining whether the stock size and fishing mortality estimated by XSA are “true” is outside the scope of this study. Therefore, the results provided by annual pseudo-stock VPA (VIT) were compared to the annual XSA outputs. This procedure allowed to quantify the relative error (bias) of each indicator. Comparison of sources of bias other than  $F_{ter}$  (e.g. associated to the definition of life-history parameters) was out of the study scope since the input data were assumed to be known (i.e. official stock assessment performed by XSA did not include uncertainty in estimates of life-history parameters). Furthermore, the impact of biased estimates of the life-history parameters on the spawning stock biomass ( $SSB$ ) derived from VIT was explored. Thus, we evaluated the variation of the  $SSB$  estimates when life-history parameters varied a proportion of 0.05, 0.10, 0.15, 0.20, 0.25 and 0.30. For performing the sensitivity analysis, we assume that a model parameterization by VIT as indicated in Table 4 and  $F_{ter}=0.5$  produced unbiased estimates of  $SSB$ .

## Results

### Estimating fishing mortality and stock size

The landings of hake in the Northwestern Mediterranean decreased by 30% from the period 2005-2009 ( $\sim 4200 \text{ t yr}^{-1}$ ) to the period 2010-2014 ( $\sim 3000 \text{ t yr}^{-1}$ ). This has been accompanied by a substantial decrease of the oldest (age class 4 and older) and youngest (class 0) individuals. Since 2010 the bottom trawler fleet was mainly targeting classes 1 and 2 and low, but persistent, removals of class 3 (Fig. 5).

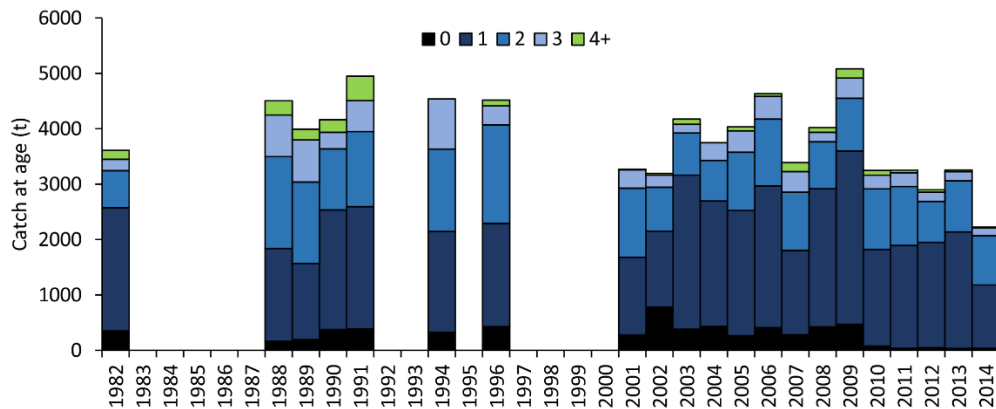


Fig. 5. Catch at age of hake caught by the bottom trawl fleet in Northwest Mediterranean Sea (data from STECF, 2015).

$F$ -at-age estimated by VIT followed the trend and held similar absolute values of those estimated by XSA (Fig. 6). Uncertainty in  $F_{ter}$  had higher impact on estimates of older ages. Estimates of  $F$  at age for ages 0, 1 and 2 by VIT were quite similar in seven of ten compared years, while  $F$  at age 3 showed larger differences regarding estimates derived from XSA. VIT produced closer values to XSA for the most fished ages (1 and 2), showing a relative error lower than 0.1 (Fig. 7a). Conversely, fishing mortality estimated by VIT in ages 0 and 3 produced larger relative errors ( $\sim 0.3$ ).

The official assessment performed by XSA considered ages 0-3 as the most fished ages ( $Fbar$ ). VIT was capable to identify the fishing mortality peak in 2006 as well as of tracking a rising trend of  $Fbar$  from 2007 to 2013 as XSA found (Fig. 8). However, VIT did not fit the absolute values of  $Fbar[0-3]$ , while the uncertainty derived from  $F_{ter}$  usually did not include the annual estimates obtained by XSA. Similar results were found when other  $Fbar$  were considered (Fig. 8). VIT failed to estimate the annual  $Fbar$  values, tending to underestimate the fishing mortality. Nevertheless, VIT could properly identify

that  $F_{bar}$  tended to increase between 2005 and 2013, as reflected by a relative error lower than 0.2 (Fig. 7b).

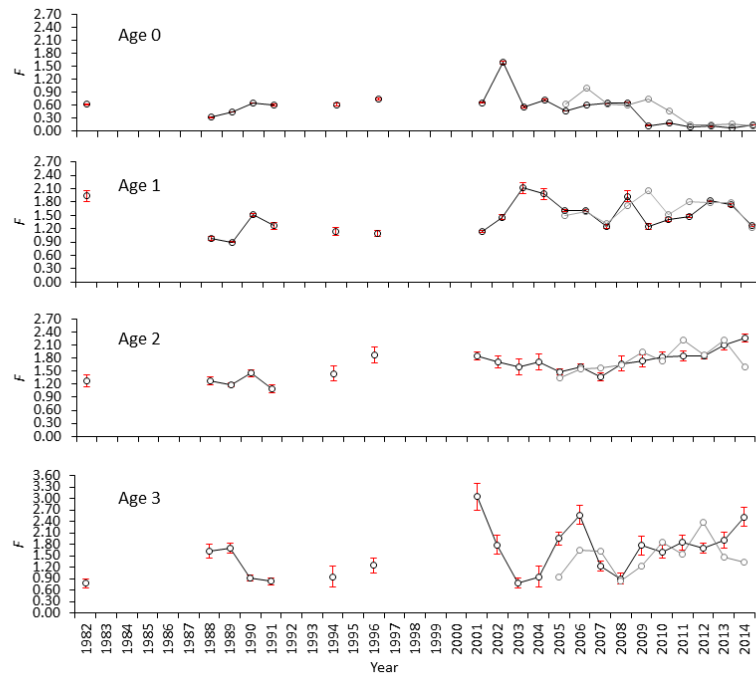


Fig. 6.  $F$ -at-age of hake produced by the bottom trawl fleet in Northwestern Mediterranean Sea. Estimations are provided by VIT (median, black) and XSA (as reported in official assessment, gray). A confidence interval of 95% derived from uncertainty in  $F_{ter}$  is shown (red bars).

$F_{bar}$  [1-3] estimated by VIT (mean= 1.60) was remarkably similar to that estimated by XSA (mean= 1.63). The  $F_{0.1}$  derived from the VIT analysis (mean= 0.31) was higher than estimated by the conventional VPA (mean= 0.26). Therefore, the overfishing level ( $F/F_{0.1}$ ) was slightly underestimated by VIT. Although relative error was equal or lower than 0.3 regardless ages computed for estimating  $F/F_{0.1}$ , more comparable values were obtained when age 0 was included (Fig. 7c).

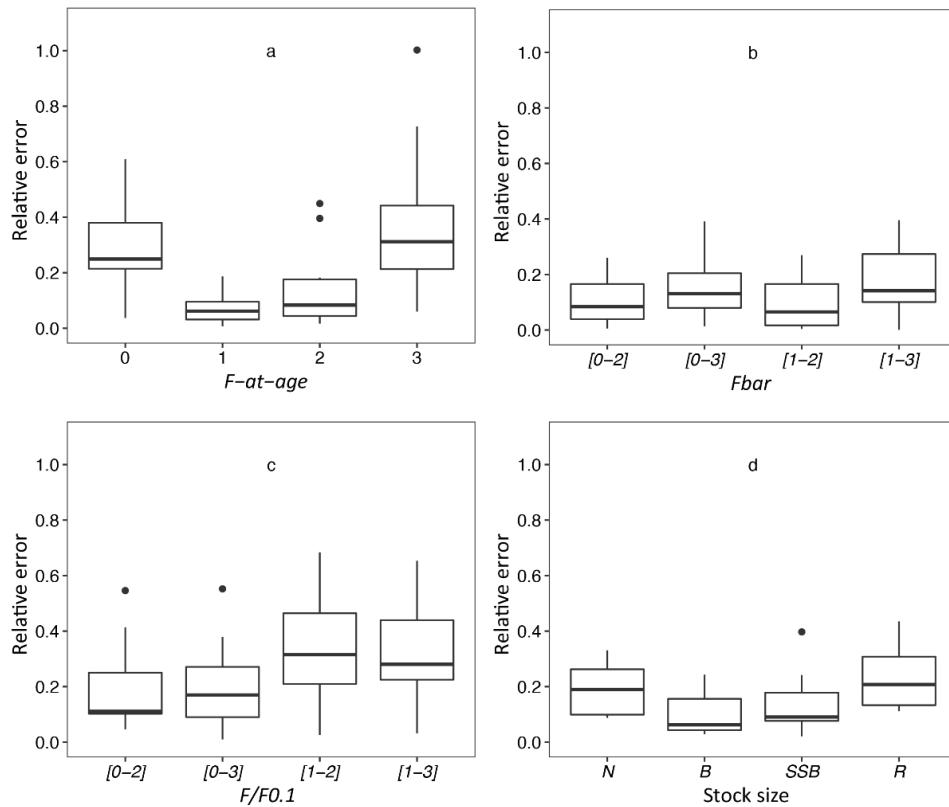


Fig. 7. Relative error in estimates of the stock indicators and reference points obtained by VIT (pseudo-cohort) and XSA (extended survivor analysis).

Trends in Stock ( $N$ ) and recruits ( $R$ ) numbers derived from by VIT assessment properly followed those estimated by XSA, although absolute values tended to be underestimated (Fig. 9). Uncertainty derived from  $F_{ter}$  was not translated into coincidence of two VPA estimates (under a confidence interval of 95%), meaning that equal parameterizing of two models did not lead fully comparable results.

Estimates of  $B$  and  $SSB$  by VIT were quite similar to those estimated by XSA, while the confidence interval always overlapped the data-rich VPA estimates. Despite the trend of the stock size showed remarkable agreement between methods, the data-limited method ranged relative errors of  $N$  and  $R$  around 0.2 (Fig. 7d). These deviations were propagated but exhibited lower variation over stock biomass and spawning stock biomass (median < 0.1).

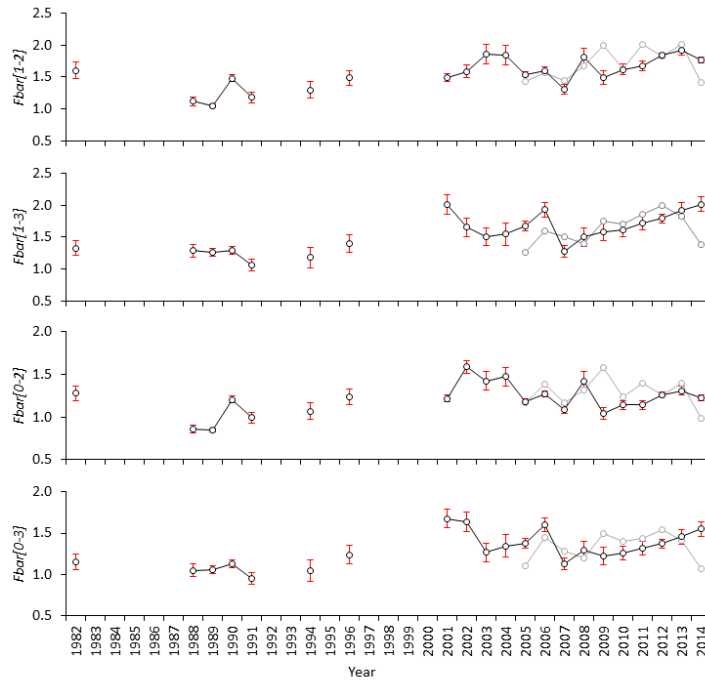


Fig. 8. Most fished ages ( $F_{bar}$ ) by the bottom trawl fleet in Northwestern Mediterranean Sea, estimated by VIT (median, black) and XSA (as reported in STECF 2015, gray). Confidence interval of 95% derived from uncertainty in  $F_{Ter}$  is shown (red bars).

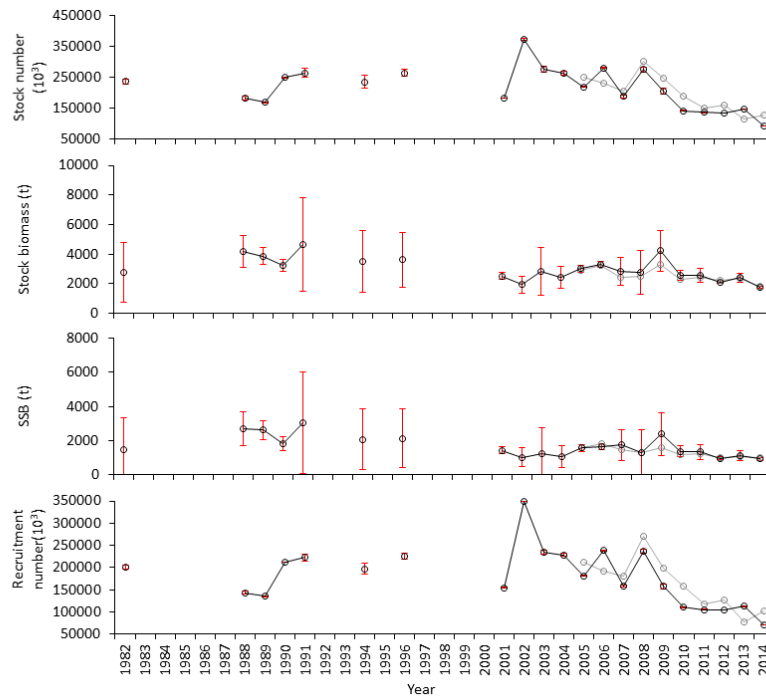


Fig. 9. Stock indicators of the European hake derived from the bottom trawl fleet catches in the Northwestern Mediterranean Sea. Estimations provided by VIT (median, black) and XSA (as reported STECF 2015, gray). Confidence interval of 95% derived from uncertainty in  $F_{Ter}$  is shown (red bars).

## Exploring the historical indicators of hake stock

Fishing mortality has tended to remain constant since 1982. The most harvested ages were 1 ( $F \sim 1.3$ ) and 2 ( $F \sim 1.5$ ), while age 3 evidenced large annual variations (Fig. 6). According to findings obtained from comparison of the VIT and XSA outputs (2005-2014), we expect that fishing mortality of ages 1 and 2 from 1982 to 2001 had been able to provide better estimates than those obtained for ages 0 and 3. Accordingly,  $Fbar[1-2]$  showed an increased trend in fishing mortality from 1988 to 2013 (also showed by  $Fbar[1-3]$  and  $Fbar[0-3]$ ), suggesting that fishing mortality has not effectively decreased along four decades (Fig. 8). The stock size ( $N$  and  $R$ ) estimated by VIT may be underestimated around 20%. Therefore, the stock number and recruits in last years (2010-2014) may have declined over 52% and 58% regarding levels estimated in the eighties and nineties (Fig. 9).

## VIT performance under bias in life-history parameters

Estimates of  $SSB$  increased between 1% and 16% when  $k$  and  $M$  (positively correlated) were overestimated by 30%. When  $L_{\infty}$  was underestimated by 30% (negatively correlated with  $k$ )  $SSB$  decreased the same proportion (Table 6). Opposite variations of these parameters produced a reduction in estimates of  $SSB$  lower than 40%. These results indicated that under low uncertainty in estimates of the life-history parameters (<15%),  $SSB$  varied less than 20% compared to “known” value. However, uncertainty larger than 20% of  $L_{\infty}$ ,  $k$  and  $M$  produced high bias in estimate of  $SSB$  (> 38%).

Misspecification of life-history parameters (i.e. without considering the underlying relationship between parameters) may lead to large bias in stock size. Erroneously setting at least one of these three parameters usually produced large variation in the  $SSB$  estimates, even if this misspecification was low (<10%). For instance, an increase of the 20% in  $L_{\infty}$  resulted in an overestimation of 148% of  $SSB$  (Table 6).

Table 6. Impact of the uncertainty in estimates of asymptotic length ( $L_{\infty}$ ), constant growth rate ( $k$ ) and natural mortality ( $M$ ) on the relative value of spawning stock biomass of European hake in Mediterranean Sea in the pseudo-cohort analyses. Change of the  $SSB$  level is measured as increase (blue) or decrease (red) when the life-history parameters are above (>) or below (<) the “true” value. As the color is darker the  $SSB$  suffered larger changes compared to the “true” value.

Kind of bias	Parameter	variation	0.05	0.1	0.15	0.2	0.25	0.3
Uncertainty in estimates of life-history parameters	$k, M$	<,<	-0.04	-0.09	-0.14	-0.19	-0.24	-0.3
	$k, M$	>,>	0.04	0.07	0.1	0.12	0.14	0.16
	$L_{\infty}, k$	<,>	0.04	0.06	0.07	0.06	0.04	0.01
	$L_{\infty}, k$	>,<	-0.05	-0.11	-0.18	-0.25	-0.33	-0.41
	$L_{\infty}, k, M$	<,>,>	-0.11	-0.22	-0.33	-0.43	-0.52	-0.61
	$L_{\infty}, k, M$	>,<,<	0.11	0.22	0.32	0.41	0.49	0.55
wrong setting of life-history parameters	$M$	<	0.11	0.22	0.34	0.46	0.6	0.73
	$M$	>	-0.1	-0.2	-0.29	-0.38	-0.46	-0.54
	$L_{\infty}$	<	-0.29	-0.55	-0.78	-0.98	-1.16	-1.32
	$L_{\infty}$	>	0.32	0.67	1.06	1.48	1.94	2.44
	$k$	<	-0.21	-0.42	-0.61	-0.79	-0.96	-1.11
	$k$	>	0.23	0.46	0.71	0.97	1.24	1.51
	$k, M$	<,>	-0.17	-0.31	-0.43	-0.54	-0.63	-0.7
	$k, M$	>,<	0.19	0.41	0.66	0.94	1.26	1.61
	$L_{\infty}, k$	<,<	-0.24	-0.42	-0.57	-0.69	-0.78	-0.85
	$L_{\infty}, k$	>,>	0.29	0.64	1.06	1.57	2.17	2.87
	$L_{\infty}, k, M$	<,<,<	-0.18	-0.34	-0.47	-0.59	-0.68	-0.76
	$L_{\infty}, k, M$	<,>,<	0.02	0.02	0.01	-0.02	-0.06	-0.12
	$L_{\infty}, k, M$	>,<,>	-0.03	-0.08	-0.14	-0.2	-0.27	-0.35
	$L_{\infty}, k, M$	>,>,<	0.38	0.88	1.53	2.37	3.44	4.79
	$L_{\infty}, k, M$	<,<,>	-0.29	-0.5	-0.65	-0.77	-0.84	-0.9
$L_{\infty}, k, M$	>,>,>	0.2	0.43	0.68	0.95	1.25	1.57	

### Discussion

We examined the usefulness of the pseudo-cohort VPA for providing estimates of conventional stock indicators in data-limited situations where long and continuous time series are not available. For this purpose, we treated the European hake as data-limited species which was assessed using VIT software and the stock assessment outputs were compared to those derived from the XSA. Our major findings revealed a remarkable agreement on the trend of fishing mortality and stock size between both methods. Fishing mortality tended to keep constant, or even increase for most of assessed years, and stock depletion was observed since the 1980s.

General agreement between annual outputs of VIT model and XSA suggests that the violation of equilibrium conditions which might have occurred during the study

period did not affect negatively the performance of VIT. Previous stock assessments based on VIT approach did not evaluate the impact of uncertainty of  $F_{ter}$  on estimates of stock indicators. Conversely, a single  $F_{ter}$  value that starts the standard VPA calculations resulted from fits of the best convergence through obtaining stable trends in the older part of the  $F$  vector (Demestre and Lleonart 1993; Aldebert and Recasens 1996). This procedure, however, could involve different unquantified bias in estimates of stock indicators among years, largely undesirable when single-year stock assessment are performed under equilibrium assumptions.

Tuning of conventional VPA has been recognized as problematic (Shepherd 1999). Thus, XSA overcomes the error-prone projections through the final year derived from tuning trials by expressing the past population abundance in terms of survivors (Shepherd and Pope 2002). European hake assessed by pseudo-cohort analysis evidenced that  $F_{ter}$  led to an unequal confidence interval in stock size indicators among years. Particularly, years when a slightly larger contribution of smaller individuals to the stock structure was observed, the uncertainty in stock size estimates considerably increased. This means that a single  $F_{ter}$  value equally used throughout temporal analysis could lead to unrealistic estimates in some years, probably related to a breach of steady-state assumptions rather than true changes in the stock size.

Fishing mortality evidenced annual biases (compared to XSA outputs) beyond of issues related to set  $F_{ter}$ . As fishing mortality is used for estimating the number at age, the stock and recruits number derived from this data-limited approach were biased over similar proportions. Fishing mortality was usually underestimated by VIT, matching with the results for the Atlantic cod when stock size was low and overexploitation was high (Rätz et al. 2010). These results suggest that despite the stock was not largely deviated from steady state, even small changes in either recruitment or fishing effort could promote underestimation of fishing mortality.

Interestingly, absolute values of fishing mortality at age for European hake were unequally underestimated. Better estimates of the most fished ages (1 and 2) and larger biases on estimates for less-represented ages (0 and 3) were found. This result suggests that the  $vBGP$  used in the official assessment was not equally well fitted to the all-years length structures, given that most of the removed individuals were found in a narrow

range (10-30 cm). Thus, fishing mortality of marginal although commonly harvested ages may be unequally observed among years due to the slicing effect. Consequently, the pseudo-cohort VPA provided less biased  $F_{bar}$  when only the most fished ages are considered (ages 1 and 2).

Estimates of  $B$  and  $SSB$  from pseudo-cohort analysis were close to those estimated by XSA, likely explained by a smaller contribution of the marginally-removed ages (ages 0 and 3) when numbers at age was translated into biomass. In fact, comparable values of  $B$  and  $SSB$  were found for most of years, unlike persistent underestimation of these indicators when the same two assessment methods were compared for the Atlantic cod (Rätz et al. 2010). Larger differences found by these authors could be due to the larger amount of age classes in cod, where full VPA methods are better suited to track cohorts over time.

A single value of  $F_{0.1}$  is derived for the whole time series by the data-rich assessment (STECF 2015, 2016), while the VIT software produces a  $F_{0.1}$  value each year, because analysis are based on pseudo cohorts.  $F_{0.1}$  computed for larger ages of the European hake led to higher annual biases between methods, as consequence of emphasizing the differences in estimates of fishing mortality for older classes. The ratio of fishing mortality to  $F_{0.1}$  produced smaller differences between estimates of both methods, offering median relative errors lower than 0.3 regardless the ages computed. A more stable annual definition of stock status by  $F/F_{0.1}$  was also recommended for data-limited stocks by Kokkalis *et al.* (2017), arguing it removes the uncertainty on growth estimation.

After gaining a better understanding of the VIT performance, our results suggested that the stock biomass of the European hake fell by half (mean) between the eighties and last assessed years. However, the unfished stock of this species could be still higher since Bas *et al.* (1955) already warned about an unsustainably fishery pressure on the European hake in the fifties. At the same time, there are evidences to support that fishing mortality has not decreased since eighties.

We conclude that pseudo-cohort VPA may offer useful information on the trend of both fishing mortality and stock size for European hake since eighties. We found it

valuable given that the twelve stock indicators and four reference points here used allowed to determine the stock status of European hake as a data-limited resource under comparable indicators used for assessing data-rich resources. Further, the comparison of the VIT model and XSA allowed to recognize the bias associated to estimates of stock indicators of European hake under steady-state assumptions.

Changes on the historical trend of fishing mortality for this species was better represented through the most fished ages (1 and 2), and the historical trend in the stock size reflected that the stock has been declining since 1982. However, the comparison between VIT and XSA evidenced that the pseudo-cohort approach was not appropriated to provide either an accurate knowledge on year-to-year variation in fishing mortality nor quantitative information on the stock size of European hake.

Data-limited species usually lack knowledge on life-history parameters (Hordyk et al. 2015; Prince et al. 2015) while the VIT software showed being quite sensitive to misspecification of  $vBGP$  and  $M$ . This implies that stock assessment based on VPA methods, including the pseudo-cohort VPA; requires a deep knowledge of both life-history parameters and harvest history. That, however, is not usually met by data-limited species. In such a situation it is essential to consider as much uncertainty as possible when setting the model, particularly when historical fishery of data-limited species is poorly known and formal assessments have not been previously performed (Ramírez et al. 2017).

In conclusion, our findings indicated that the pseudo-cohort VPA approach offered stock indicators trends of European hake with a remarkable agreement compared to XSA outputs. The annual bias in estimates of the assessed stock indicators and the reference points used in this study were usually lower than 20%. Therefore, the absolute values derived from pseudo-cohort VPA should not be used as basis for providing advice with management purposes. In order to interpret the temporal trend of the stock indicators that emerges by using all sparse years together, it is recommendable to recognize the impact of variation of  $F_{ter}$  by year. Thus, this additional information helps to understand whether if the observed changes, for instance, on the stock size are linked to real changes of the stock in the sea among years or are related to the guessed value of  $F_{ter}$  that was used. According to our findings, the multi-species

advice for both “target” and “secondary” species, supported by comparable stock indicators, could provide additional scientific arguments for evidencing a whole fishery impact on the removed resources. Applying VIT to sparse/discontinuous data series of length frequencies, which are available at least since the early 1980s in many Mediterranean marine research centers, would help providing a more complete picture on the trend of historical evolution of fish stocks in the area, by providing estimates of stock indicators over several decades. However, assessment methods able to consider the uncertainty in both the life-history information but fishery-dependent data should become part a deeper exploration of the stock status trend of data-limited stocks.

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2.2. European hake (*Merluccius merluccius*)  
assessment based on size frequencies and basic  
biological parameters in the SW Mediterranean



**European hake (*Merluccius merluccius*) assessment based on size frequencies  
and basic biological parameters in the SW Mediterranean**

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**Abstract**

Assessments of small-scale, coastal fisheries are often hindered by the lack of complete data sets fulfilling the requirements of conventional stock assessment methods. In contrast, existing information from the regular monitoring of fisheries coupled with data-limited assessment methods can help provide the information needed for effective management of fisheries. We investigated the year-to-year length-based estimates of the spawning potential ratio (SPR) of European hake, a species with moderate data availability (i.e., reasonable knowledge of the life-history traits and temporal length structures of the population). We explore the effects of considering fast- vs. slow-growth hypotheses and the uncertainties of input parameters on the annual estimates of the SPR (1998-2010) in a bottom trawl fishery in the SW Mediterranean (Murcia region). The SPR estimates under four life-history scenarios were sensitive to the quality (sample size) of the length frequencies. Consequently, comparable and more accurate SPR estimates were often found for annual samples larger than 2,000 individuals. The  $F/M$  ratio varied less when size compositions were

grouped by periods, suggesting there are transitory-population size structures. The SPR estimates were insensitive to the underlying growth hypothesis since similar  $M/k$  ratios  $>2.3$  were derived from the four sets of von Bertalanffy growth parameters. The assessment indicated overexploitation of hake ( $F/M > 2$ ,  $SPR < 10\%$ ) that is comparable with that derived from conventional assessments for this species elsewhere in the Mediterranean Sea. According to our findings, the LB-SPR method can provide reliable stock assessments and allows population trends to be calculated for data-limited species. However, this method requires a good understanding of the input data and their possible sources of bias.

Keywords: European hake; spawning potential; fisheries assessment; SW Mediterranean; trawl fisheries.

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### **Introduction**

Stock statuses of target fishery resources have conventionally been estimated using virtual population analyses (Hilborn and Walters, 1992; Lassen and Medley, 2001) and statistical catch-at-age models (Methot and Wetzel, 2013; Scott et al., 2016). Information on life-history parameters is consequently expected to be reasonably well-known, and the volume of landings and size distribution of catches are expected to be properly monitored at least twice during the species lifespan. Fishery-independent data are also used to provide abundance/biomass indices. The fishery resources that meet these requirements are commonly categorized as “data-rich” species (Kokkalis et al., 2015).

However, many target species, as well as most stocks in multispecies fisheries, do not meet the requirements for assessment using data-rich methods. These species, by default, are usually considered data limited. The assumptions underlying the methods to assess data-limited fisheries can circumvent the lack of fishery information (e.g., catch-at-age). However, defining, for example, how much the fishing mortality is

above or below a reference point requires appropriate knowledge of life-history traits (Kokkalis et al., 2015; Magnusson and Hilborn, 2007; Ramírez et al., 2017).

The spawning potential ratio (SPR; the ratio of the reproductive potential of a stock subject to a given amount of fishing pressure to that of an unfished stock; Goodyear, 1993) can be estimated from length-based techniques using data-limited methods (Hordyk et al., 2015c). SPR ranges between 0 and 1, with a value of 1 representing an unexploited stock. Satisfactory snapshots of SPR estimates have been obtained for species with little or no knowledge of their life histories (Prince, 2017; Prince et al., 2015). The performance of this method depends on the accurate estimates of life history ratios, and it is especially sensitive to the estimates of  $L_{\infty}$  (Hordyk et al., 2015c). In addition, Prince et al. (2015) found that the most uncertain assessments were derived from small size samples. Accordingly, the uncertainties of length-based spawning potential ratio (LB-SPR) outputs were quantified by these authors by considering plausible values of life-history parameters and resampling the available length data.

Hake (*Merluccius merluccius*) is one of the most important European target species of bottom trawl nets and set longlines, and annual catches of ca. 25,000 t have been reported in recent years from Mediterranean waters (FAO, 2011). According to conventional stock assessments, this species is overexploited in all Mediterranean fishing subareas in recent decades (Cardinale and Scarcella, 2017; Colloca et al., 2013). However, unsustainable hake harvest was already a concern before that, at least for northwest Mediterranean stocks (Aldebert et al., 1993; Bas et al., 1955; Oliver, 1994).

The stock assessments of hake in the northwest and southwest Mediterranean Sea by working groups of the STECF (Scientific, Technical and Economic Committee for Fisheries) or the GFCM (General Fisheries Commission for the Mediterranean) in recent years have relied on the fast-growth hypothesis that was principally justified from a tag-recapture experiment in the Gulf of Lions (Mellon-Duval et al., 2010) which determined two different growth curves when females (model 1) or females, males and undetermined sexes (model 3) were considered. On the other hand, the slow-growth hypothesis of hake was considered to be appropriate in early assessments (Aldebert and Recasens, 1996) and is supported by this study (see below). Consequently, we examined

the sensitivity of the results of the LB-SPR method to a range of  $L_{\infty}$  and  $k$  values from 100.7 to 110 cm and from 0.112 to 0.236  $\text{y}^{-1}$ , respectively (“slow” and “fast” growth hypotheses).

Reasonable bounding estimates of the von Bertalanffy growth parameters (*vBGP*) and natural mortality (*M*) of the hake stock in the Mediterranean Sea and a comprehensive set of length frequencies recorded onboard a commercial fleet from 1998 to 2010 in the Murcia region (southwest Mediterranean Sea) provide high-quality data for estimating year-to-year length-based SPR values. In this sense, hake will be considered as a species with moderate data, assessed using a data-limited framework. Therefore, the objective of this work is to assess the exploitation rate of hake in the SW Mediterranean based on a size-frequency analysis of individuals sampled from commercial catches over 13 years using the LB-SPR method under different assumptions and uncertainties of life-history parameters. We also examine whether the results from the LB-SPT method deviate from those of conventional assessment methods.

### **Materials and methods**

In the Murcia region, as elsewhere in the Mediterranean, hake is of high economic importance for the trawl fleet. Catches for 1998 – 2010 fluctuated from 208 t in 2000 to 60 t in 2008, with an average catch of 109 t (Table 7). The trawl fleet has declined continuously in number from 39 in 2000 to 29 in 2010. The earliest formal stock assessment in the Murcia region, corresponding to data between July 1998 and June 1999, showed strong overexploitation of the species (Martínez-Baños, 2000), which has continued, as evidenced in the recent stock assessments of subarea 1 by the GCFM, to which the Murcia region belongs (STECF, 2015, 2011).

Table 7. Landings (tonnes) and numbers of measured individual hake fished by trawlers during 1998-2010 in the Murcia region (SW Mediterranean, Spain).

Year	Landings (t)	Total individuals
1998	111.67	4436
1999	102.541	4505
2000	208.23	4199
2001	184.34	2697
2002	123.334	1041
2003	83.942	1990
2004	93.973	2231
2005	66.903	1691
2006	109.724	2985
2007	89.571	1416
2008	59.992	1608
2009	109.105	1594
2010	76.1	1948

#### Length frequency data

Hake, along with other demersal species targets of the trawl fishery, are regularly monitored in the Murcia region by the local fisheries department of the Autonomous Government, in the ports of Mazarrón, Águilas and Cartagena (Fig. 10).

Hake were measured onboard commercial trawlers working on hake fishing grounds (50-350 m depth approximately) in the Murcia region (Fig. 10) from April to December 1998 to 2010, from 85 fishing trips. Two or three fishing trips were sampled each month over three depth strata: stratum A 50 to 150 m, stratum B 150 and 350 m, and stratum C deeper than 350 m. The sampled data were weighted by catch-in-number by depth stratum and month to obtain a representative annual size distribution. Thus, the contribution of the sampled individuals to the size structure of the stock caught by the bottom trawl was proportional to the total individuals harvested by the fleet every year.

Fish brought onboard were randomly sampled across all commercial categories (small, medium and large). Individual sizes (cm TL) were measured to the lowest half centimetre. We measured 31,969 hake with sizes ranging between 5 and 74 cm TL. The modal value was 16.5 cm TL, and 95% of individuals were between 7 and 37.5 cm TL. Length frequency data after 2010 were not considered because of regulatory mesh

changes in Mediterranean trawls (adoption of 40-mm square meshes) that could imply changes in catchability. Hake is not fished with fishing gear other than bottom trawls in the Murcia region (i.e., set longlines or gillnets are not employed).

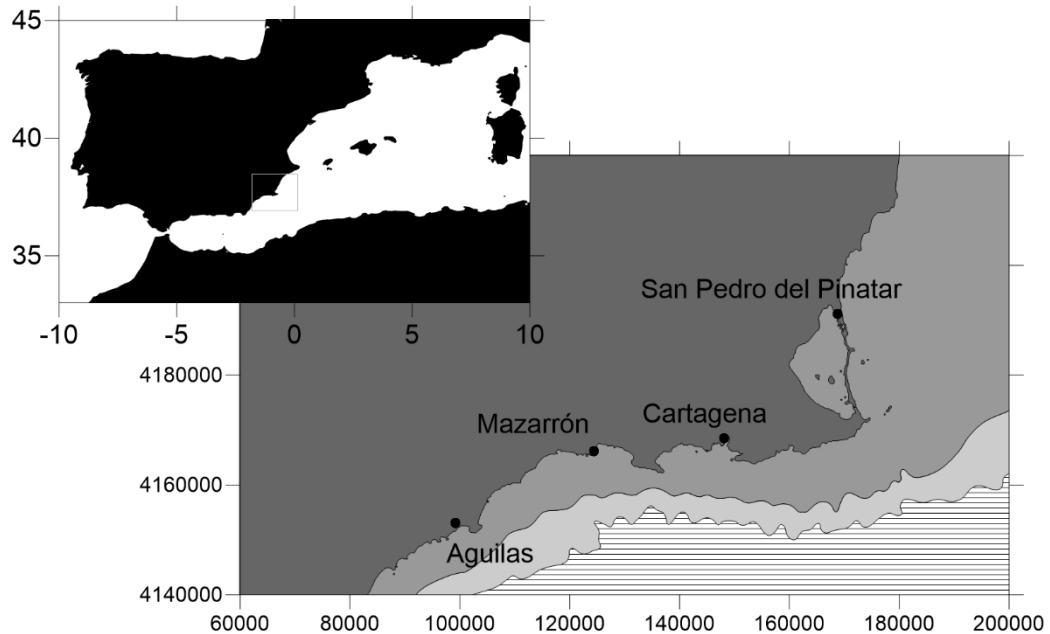


Fig. 10. Study area (Murcia Region) showing harbours where the bottom trawl fleet landings were monitored. Depth contours shown are 200 and 1000 m.

Despite the fact that the trawl gear is often assumed to catch a wide range of sizes, some studies have indicated a dome-shaped selectivity pattern (Binion et al., 2009; Kuparinen et al., 2009). This pattern occurs because the size distribution perceived by trawl samples may be caused by ontogenetic migrations to deep waters (Jacobson et al., 2001). In particular, this behaviour has been associated with the European hake in the West Mediterranean (Abella et al., 1997; Caddy, 2015). In addition, overrepresentation of some sizes in trawl catches may occur depending on the growth hypothesis for the species (Huse et al., 1999). Consequently, the asymptotic selectivity assumed by the LB-SPR model (Prince et al., 2015) may not be totally valid, requiring careful interpretation of the SPR results for European hake.

## Life-history parameters

We used four sets of von Bertalanffy growth parameters, three taken from the literature and the fourth estimated in this study from the raised annual size frequency. There are two contrasting hypotheses regarding hake growth in the Mediterranean, with  $k$  values of approximately  $0.1 \text{ yr}^{-1}$  for the slow-growth hypothesis and  $k$  values of approximately  $0.2 \text{ yr}^{-1}$  for the fast-growth hypothesis (Table 8). We used models 1 and 3 of Mellon-Duval et al. (2010), which were derived for a hake stock in the Gulf of Lions as representatives of the “fast growth” hypothesis. The two parameter sets representing the “slow growth” hypothesis (Table 8) were obtained from Aldebert and Recasens (1996, females) and from the length frequencies used in this study (Martínez-Baños et al. (Data in Brief)), pooling all years and using the MIX model (Macdonald and Green, 1988; software MIX R version: library mixdist v. 0.5-4 in R v. 3.0.2). The results obtained for the size-age distribution of hake in the Murcia region were  $L_{\infty} = 105.874 \text{ cm TL}$ ,  $k = 0.112 \text{ yr}^{-1}$  and  $t_0 = -0.167 \text{ yr}$ .

Table 8. von Bertalanffy growth parameters and natural mortality of hake in the Mediterranean Sea used in this study.  $M$  was estimated using PRODBIOM from the average  $M$  at ages 0-5 (Abella et al., 1997).

Source	Region	Sex	$L_{\infty}$ (cm)	$M$ ( $\text{yr}^{-1}$ )	$K$ ( $\text{yr}^{-1}$ )	Growth hypothesis	$M/k$
Mellon-Duval et al., 2010, model 3	Gulf of Lions	Male, female, undetermined	110	0.565	0.178	Fast	3.17
Mellon-Duval et al., 2010, model 1	Gulf of Lions	Female	100.7	0.62	0.236	Fast	2.63
Aldebert and Recasens, 1996	Catalan Sea	Female	100.7	0.29	0.124	Slow	2.34
Martínez-Baños et al. (Data in Brief)	Murcia	Combined	105.87	0.34	0.112	Slow	3.04

The LB-SPR method assumes constant natural mortality ( $M$ ) for all recruited size classes. However, the stock assessment of hake in the Mediterranean Sea uses an  $M$ -at-age vector (1.24, 0.58, 0.45, 0.40, and 0.37 for ages 0 to 5+) (STECF, 2015), which was estimated using the PRODBIOM indirect estimator (Abella et al., 1997). This estimator is frequently used to estimate  $M$  for Mediterranean resources. In particular, this  $M$  estimator supports the “refuge” concept for older hake ages, which was considered by Abella et al. (1997) as being partially responsible for the sustained productivity of fisheries harvesting the juvenile fraction of stocks. Given that PRODBIOM uses values for  $L_{\infty}$  and  $k$ , along with other information to estimate  $M$ , each set of von Bertalanffy

growth parameters produced a different  $M$ . Model 3 (Mellon-Duval et al., 2010) is used in the official stock assessment (STECF, 2011). The  $M$  derived from the abovementioned estimator met the considerations made by ICES (2014), who advised that a fast-growth hypothesis for hake will imply an  $M$  value that is nearly double that implied by the slow-growth hypothesis (Table 8).

#### LB-SPR method

The method developed by Hordyk et al. (2015a) calculates SPR for a stock directly from the size composition of the catch. It requires the size composition data of the catch as input data as well as a minimum set of biological parameters:  $L_{\infty}$ , the coefficient of variation (CV) of  $L_{\infty}$ ,  $M/k$  and the size of maturity. The four  $L_{\infty}$  parameters and the ratios between natural mortality and the von Bertalanffy growth rate ( $M/k$ ) were estimated (Table 8).

Length observations were abundant each year ( $n > 1,000$ , Table 1), but ageing was not performed. According to Prince et al. (2015), the CV of  $L_{\infty}$  can be assumed to be approximately 10% when reliable length and age data are not available. We assumed that this value was valid to including in the LB-SPR model for European hake. In addition, we explored the sensitivity to this value by considering CVs for  $L_{\infty}$  of 0.05, 0.1, 0.15, 0.2, 0.25 and 0.3, compared to a hypothetical low-variance value of CV of  $L_{\infty}=0.001$  (assumed “true” value). A boxplot was used to show the LB-SPR outputs ( $F/M$  and SPR) from 52 runs for each assumed value of the CV of  $L_{\infty}$ .

The sizes at which 50% ( $L_{50} = 30$  cm) and 95% ( $L_{95} = 40$  cm) of the population is mature were obtained from the raw data used for the official stock assessment of hake (STECF, 2015). From the life-history ratios ( $M/k$ ,  $L_{50}/L_{\infty}$  and  $L_{95}/L_{\infty}$ ), the model allows the estimation of the selectivity at length ( $L_{s50}$  and  $L_{s95}$ ) and the exploitation ratio  $F/M$ , which are used to calculate the SPR (see Hordyk et al., 2015c for a full description of the model).

### Uncertainty of life-history parameters

Additional uncertainties in the SPR outputs were quantified by placing distributions bounded by upper and lower values of  $L_\infty$ , the CV of  $L_\infty$ ,  $M/k$ ,  $L50$  and  $L95$  to account for the uncertainties of these parameters. In addition to the R library, “LBSPR” used to estimate the SPR for each year, the R library “triangle” was used to generate the life-history parameters.

The upper value of the mean asymptotic length ( $L_\infty$ ) was 115 cm because the hake population in the Murcia region, according to the current evidence, is largely truncated. A maximum value of 110 cm could underestimate  $L_\infty$  for an underexploited population. We selected 105 cm as the mean  $L_\infty$  value, given that this value corresponded to estimates from the Murcia region data (the lower  $L_\infty$  was 95 cm). The lower, upper and middle values of the triangular distribution for the CV of  $L_\infty$  were set to 0.05, 0.3 and 0.15, respectively. Considering the life-history parameters used in this study, we used  $M/k$  values from 2.2 to 3.2 (mean  $M/k = 2.75$ ). Given that the estimation of the asymptotic length of hake may be subject to considerable uncertainty, but  $L50$  is comparatively better known, we assumed a constant  $L50/L_\infty$  ratio = 0.28. We used 10,000 random draws to obtain the distributions of the life-history parameters and the final distribution of the LB-SPR estimates per year.

The population may be far from equilibrium, implying that size composition is affected by the transitory dynamics of mortality and recruitment. Assuming that the length frequency data are the only available information, we explored the  $F/M$  and SPR estimates by grouping the annual length frequencies in four periods (1998-2000, 2001-2003, 2004-2006 and 2007-2010). Thus, we provided a smoothed trend of the LB-SPR estimates to reduce the potential effect of the annual non-equilibrium state.

### Results

Fig. 11 shows the fit to the length frequency data by year for the growth parameters estimated in this study. For this set of parameters.  $Ls50$  varied around 11.5 cm TL and  $Ls95$  around 14 cm TL (Fig. 12 left panel) while  $F/M$  decreased substantially from ca. 3.0 in 2007 to ca. 2.0 in the most recent years (Fig. 12 central panel), and the SPR was estimated to less than 0.1 but to have increased in the most recent four years

(Fig. 12 right panel). The model outputs were similar for the other three life-history parameter sets in Table 8 (Figures S1, S2 and S3). The selectivity parameters remained quite constant (average  $L_{50} \sim 11.2$ ;  $L_{95} \sim 14.2$  across the four growth curves, Fig. 12, S1, S2 and S3), i.e. the values of the selectivity parameters were insensitive to the life-history ratios used as inputs to the LB-SPR model ( $CV < 0.5\%$  for both  $L_{50}$  and  $L_{95}$ , not shown).

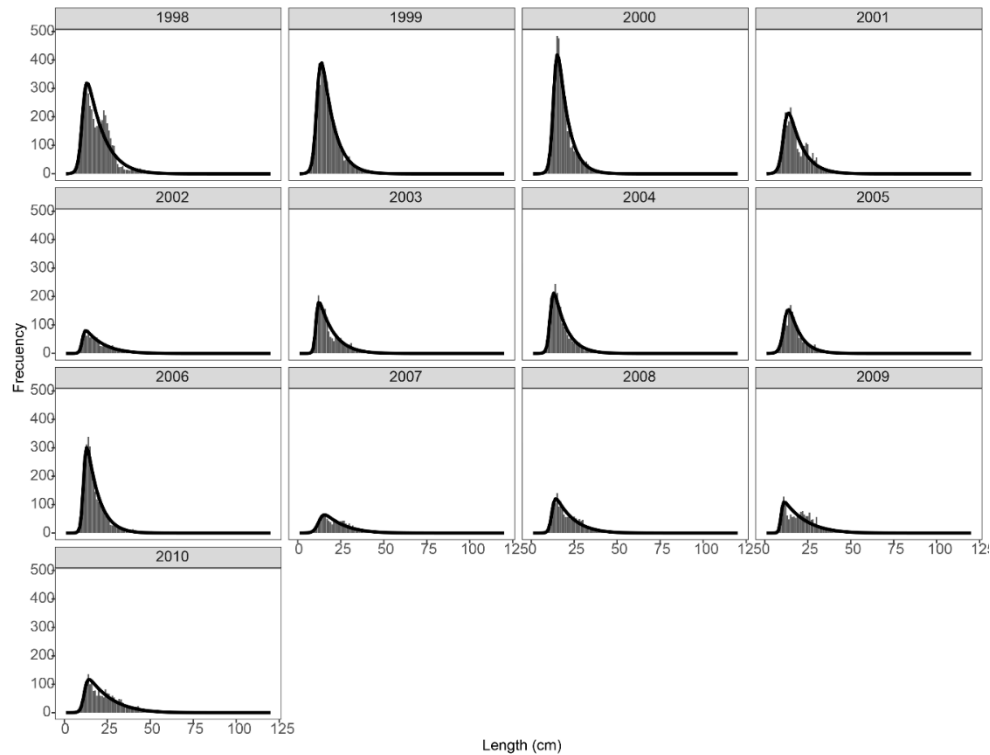


Fig. 11. Observed and LB-SPR model-predicted length-frequency histograms (bars and gray lines respectively) when the LB-SPR model was fitted using the life-history parameters derived from the Murcia region.

The highest ratio between fishing and natural mortality  $F/M$  (Fig. 4, Table S1) was for the lowest  $M/k$  value (2.34). Conversely, high  $M/k$  values (3.04 and 3.17), regardless of the growth hypothesis, led to the lowest estimates of  $F/M$ .

Although mean  $F/M$  derived from the four life-history parameters varied by  $\sim 79\%$  per year from 1998 to 2010, the corresponding SPR estimates exhibited limited changes during some years (e.g., 1999, 2004 and 2006, Fig. 13, Table S1) or were clearly differentiated in others (e.g., 2007-2010, Fig. 13, Table S1). Fig. 11 indicates that the LB-SPR model was able to fit the length-frequency distributions for 1999, 2004 and 2006 better than those for other years, with the best fits coinciding with large samples sizes

( $n > 2,000$ ) (Table 7). Conversely, many of the annual models exhibiting poor fits to the length-frequency distributions (2007-2010, Fig. 11) coincided with sample sizes less than 2,000 individuals (Table 7). Thus, a good fit to the length frequency data by year usually led to similar SPR values (Fig. 13, Table S1), while poor fits tended to produce noticeable differences in the SPR estimates. We speculate that a larger number of length samples each year contributed to reducing the differences between the SPR estimates derived from the four life-history parameters.

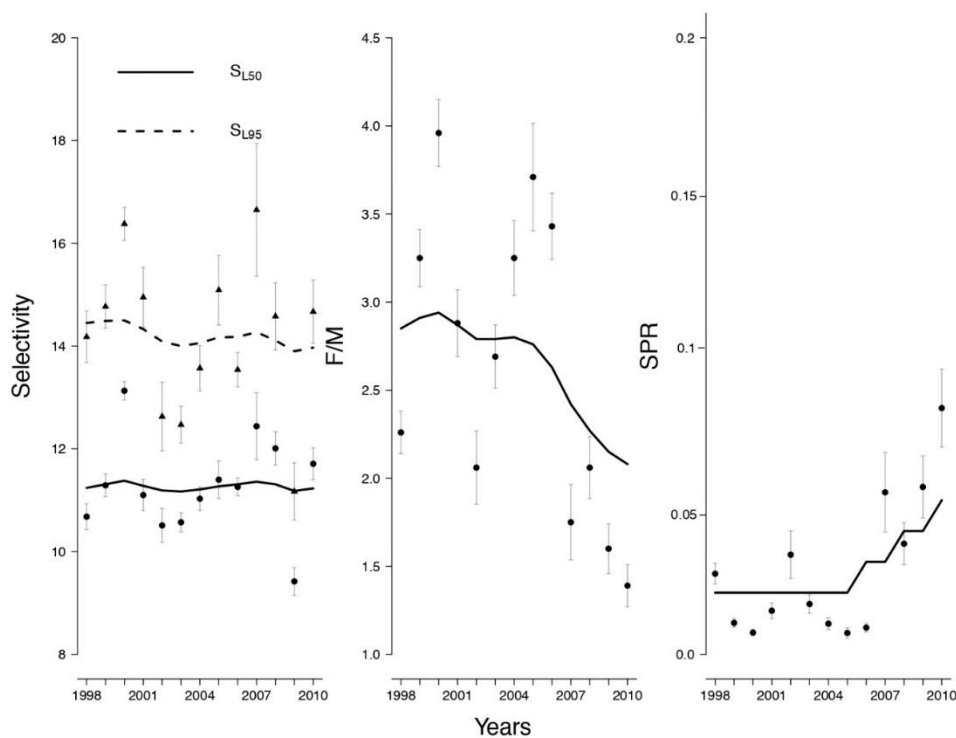


Fig. 12. LB-SPR outputs for the life-history parameters estimated for the Murcia region. Annual (individual symbol) and smoothed model (black line) estimates of selectivity,  $F/M$  and SPR. Confidence intervals (95%) are shown in vertical bars.

Poor fits for some years did not prevent the SPR estimates for the four sets of life-history parameters to be consistently below 5% from 1998 to 2006. Since 2007, the SPR tended to increase but was never higher than 8%. As for  $F/M$  estimation, the estimate of SPR did not depend on the growth hypothesis, but rather on  $M/k$ . The LB-SPR model was relatively insensitive to a CV of  $L_{\infty} < 0.15$ , but higher values caused differences in the SPR estimate that were greater than 10% (Fig. 14).

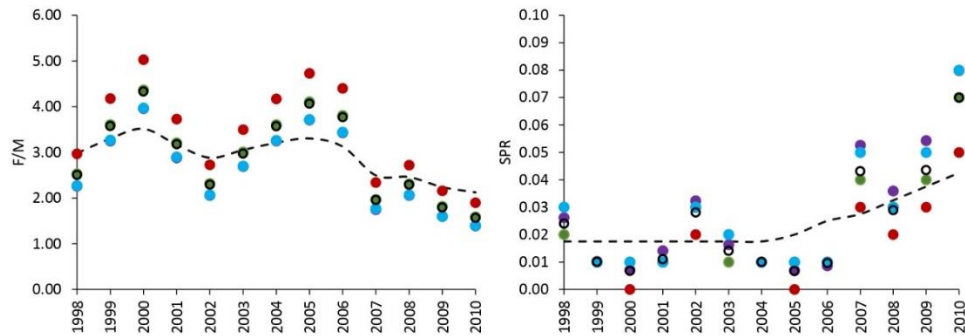


Fig. 13.  $F/M$  (left panel) and SPR (right panel) estimated using growth parameter estimated during this study (purple circles), by Aldebert and Recasens (1996) (red circles), from model 1 of Mellon-Duval et al.(2010) (green circles), and from model 3 of Mellon-Duval et al.(2010) (blue circles). Mean individual year (black empty circles) and mean smoothed model (dotted lines) estimates are shown.

The estimates of  $F/M$  and SPR were not very sensitive to the uncertainties in the life-history parameters, on year-to-year basis (Fig. 15a, b). The size frequency data grouped over three- or four-year periods substantially reduced the overall  $F/M$  ratio, which ranged from 1.5 to 4 (Fig. 15c). Thus, the relative fishing mortality estimated by the LB-SPR model decreased by more than 42% when data were grouped. However, this reduction in  $F/M$  did not translate into changes in SPR estimates (Fig. 15d), and comparable SPR values were observed for annual (Fig. 15a) and year-grouped (Fig. 15d) assessments.

## Discussion

We investigated the year-to-year variability in length-based estimates of the SPR for hake, a species with moderate data availability. We explored the effect of fast- and slow-growth hypotheses and accounted for the uncertainties of life-history parameters on the annual SPR estimates. The similarity of the annual estimates of the SPR under the four life-history parameters depended strongly on how well the model fitted the size data (Fig. 11), while the SPR estimates were not sensitive to the growth hypotheses.

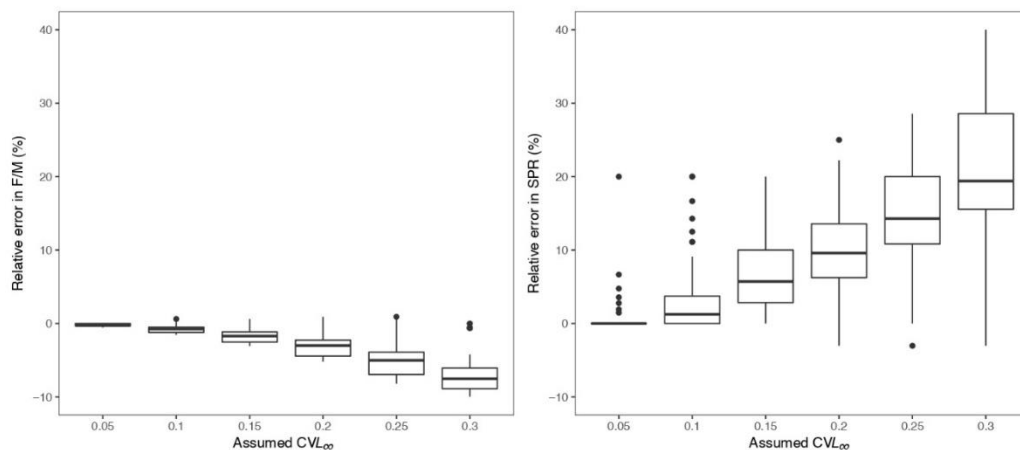


Fig. 14. Differences in estimates of  $F/M$  and  $SPR$  for various assumed values for the CV of  $L_{\infty}$  from the estimates when this CV is set to 0.001.

The fast-growth hypothesis for West Mediterranean hake (Garcia-Rodriguez and Esteban, 2002; Mellon-Duval et al., 2010) is compatible with high relative frequencies of the youngest ages (Mellon-Duval et al., 2010). Older ages from an unfished stock would be less well-represented, corresponding to stocks exhibiting an  $M/k$  value  $> 1.5$  (Hordyk et al., 2015a, b). Aldebert et al. (1993) found that the individuals landed by bottom trawl, gillnet and longline fleets in 1988 were mostly smaller than 50 cm. Although young ages have been highly fished by the bottom trawl fishery since at least the 1950s, large individuals were already rare at that time (Bas et al., 1955). Thus, it was not unexpected that individuals near asymptotic size ( $> 90$  cm) were not recorded in our study nor in previous stock assessments (Aldebert et al., 1993; STECF, 2015).

Natural mortality of hake in the Mediterranean Sea is often estimated using PRODBIOM. Abella et al. (1997) suggest that production of hake is sustained despite high trawling pressure on juveniles, by accepting the “spawning refuge” concept for older ages. PRODBIOM produced average  $M$  estimates for the fast-growth hypothesis that were approximately twice those produced for the slow-growth hypothesis, coinciding with the rationale for choosing  $M$  for hake recommended by ICES (2014). Although PRODBIOM uses several life-history parameters ( $L_{\infty}$ ,  $k$ , estimated age and the  $a$  and  $b$  values from the length-weight relationship) to estimate natural mortality, the mean  $M$  estimates are strongly related to the  $k$  value ( $r^2=0.87$ ). This result implies that the concept of the fast- or slow-growth hypothesis virtually disappears when  $M/k$  instead of

$M$  and  $k$  separately is used. Thus, two growth hypotheses produced fairly similar  $M/k$  values (Table 7).

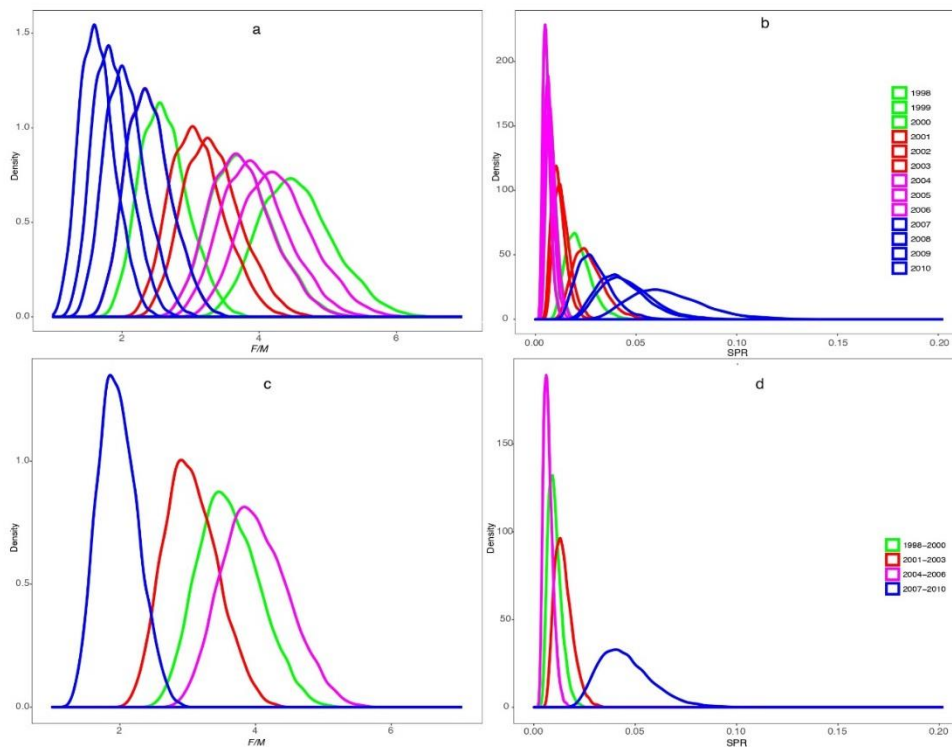


Fig. 15. Estimates of  $F/M$  (a,c) and SPR (b,d) for the European hake in the southwest Mediterranean. Results are shown by year (a,b) and when data are grouped (c,d).

Mellon-Duval et al. (2010) questioned the reliability of the early estimates of age-structure of hake in the Mediterranean after finding evidence for fast instead of slow growth. However, Bertignac and De Pontual (2007) found that a bias in estimated ages did not affect either trend in fishing mortality nor the stock status of hake in the Bay of Biscay. On the other hand, the LB-SPR method estimates the ratio of fishing mortality to natural mortality from length-composition data, which are assumed to be representative of the exploited stock. Therefore, the SPR estimate overcomes the uncertainty associated with growth, but requires the identification of appropriate  $M/k$  and  $L50/L_{\infty}$  ratios.

$M/k > 2.3$  for the hake stock in the Murcia region corresponds to unfished stocks dominated by young classes (Hordyk et al., 2015b). These  $M/k$  values may be compatible with a “natural” stock that is mainly represented by young individuals, as can be expected in a species that has faced severe overexploitation for decades. Stock assessments for this kind of population, with low representation of adult classes, would

be less sensitive to the possible bias introduced by ignoring dome-shaped selectivity (Hordyk et al., 2015c). The absence of large individuals in the size frequency would not result in large underestimates of the SPR if selectivity was not dome-shaped. However, we cannot rule out the possibility of dome-shaped selectivity, i.e., the apparent absence of large individuals from bottom trawl catches due to variations in catchability for biological reasons (Mahévas et al., 2011). Specifically, diel vertical migrations of larger individuals and/or their preferential distribution in non-trawled areas may result in dome-shaped selectivity because Spanish Mediterranean trawls are permitted to operate during daytime hours only.

As a species with moderate data availability, the asymptotic lengths of Mediterranean hake are reasonably well-known (Aldebert and Recasens, 1996; Mellon-Duval et al., 2010). The largest and lowest estimates used in this study, representing the two growth hypotheses, did not exceed 10% of the asymptotic length estimated for the Murcia region ( $L_{\infty}$  = 105.87 cm). In addition, a coefficient of variation of  $L_{\infty}$  lower than 0.3 had small impacts on the LB-SPR outputs in our study, coinciding with the findings of Hordyk et al. (2015c). Thus, the probabilities of underestimating or overestimating  $F/M$  or affecting the estimate of SPR due to inaccurate or imprecise estimates of  $L_{\infty}$  were low (Hordyk et al., 2015c).

The performance of the LB-SPR method in assessing fish species from Fiji (Hordyk et al., 2015c; Prince, 2017) compares favourably with our results. In these assessments sample sizes of ~1,000 individuals were required to produce robust estimates of the SPR. Even though our sample sizes were larger than 1,000, the SPR estimates from the four life-history parameter sets varied among years (from 1% to 6% approx.). The differences are likely insignificant since the SPR estimates are far ( $SPR < 10\%$ ) from the target level for a sustainability fishery in any case ( $SPR > 40\%$ , Myers et al., 1994). In the framework of fisheries management using a harvest control rule, even small changes to the current SPR could mean additional time to return to sustainable levels because more than one recovery trajectory may occur (Goodyear, 1993; Hordyk et al., 2015a).

The years when the four life-history sets produced virtually the same SPR values were derived from large samples (>2,000 individuals) and corresponded to most of the best fits to the annual length distributions. These results suggest that the life-history

parameters of hake that were explored in this study ( $L_{\infty} > 100\text{cm}$ ,  $0.11\text{yr}^{-1} < k < 0.24\text{yr}^{-1}$  and  $0.28\text{yr}^{-1} < M < 0.63\text{yr}^{-1}$ ) and the possibilities of either fast- or slow growth lead to similar LB-SPR estimates if the length structure properly represents the population.

Regardless which of the four life-history parameter sets was used, the SPR was never higher than 8% during the study period. Comparable SPR values were obtained accounting for the uncertainties of the estimates of  $L_{\infty}$ , the CV of  $L_{\infty}$ ,  $M/k$  and maturity-at-size. Nevertheless, the range of SPR estimates was quite different between the early and late years, indicating a relatively precise SPR lower than 2.5% before 2006 (except for in 1998 and 2002) and more variable and increasing SPRs since 2007.

Although an evaluation of whether hake in the West Mediterranean are out of equilibrium is beyond the scope of this study, hake recruitment in the West Mediterranean (GSA: 01, 05, 06 and 07, see STECF, 2015) from 2003 to 2014 showed moderate variability ( $\sigma_R = 0.33\text{-}0.41$ ). Hordyk et al. (2015c) found that the LB-SPR method performed relative well when  $\sigma_R < 0.6$ , producing estimates within 30% of the true value. This result matched the lower variations of  $F/M$  and the comparable values of the SPR estimates when the size compositions were grouped into three- or four-year periods. Thus, moderate changes in recruitment, which probably led to transitory population size structures, did not prevent the reliable SPR estimates for hake.

Most of the hake historically removed by the trawler fleet in the Murcia region were of sizes much less than the asymptotic length. In addition,  $L_{50}$  and  $L_{95}$  correspond to 37% and 47% of the length-at-maturity ( $L_{50} = 30\text{ cm}$ ), respectively. Sustained exploitation under this fishing regime likely explains why the SPR remained at or below 5%. The SPR increased to 8% towards the end of the study period despite the fact that the selectivity did not improve. In the absence of the knowledge of temporal changes of catchability, increased growth rates due to a favourable trophic environment (or decreased mortality) could be hypothesized.

A persistently very low SPR together with historical exploitation mostly concentrated on younger individuals raise an obvious question. What has prevented the collapse of the Western Mediterranean hake fishery? Caddy (2015) suggested that large individuals may be unavailable to the fishing gear and protected in “spawning refugia”.

This would imply that both conventional (STECF, 2015) and data-limited (e.g., this study) stock assessment methods would overestimate fishing mortality and underestimate spawning biomass.

A complementary interpretation of very low and persistent SPR values should be considered for management purposes. The low SPR values provide evidence of excessive fishing pressure on the hake stock, mainly the juveniles. This result may imply that the population size is at risk of decreasing even further because the refuge for older ages may not continue to operate to compensate for the current fishery pressure as occurred during early stages of overexploitation. Therefore, both reducing the fishing pressure on young ages and allowing the recovery of the spawning stock biomass is desirable. Thus, advice that is based on a reference point for juveniles (< age 2) as already suggested (Caddy, 2015; Colloca et al., 2013) makes sense for Mediterranean hake stocks.

### **Conclusions**

We estimated the SPR of hake, a species with moderate data availability (appropriate knowledge of life history traits and length frequency distributions based on large samples over several years). The LB-SPR estimates were not affected by the assumed growth hypothesis since  $M/k$  leads to similar values regardless of the assumed growth parameters. However, variations in model fits and consequently estimates of  $F/M$  and SPR depend on sample size and representation of the length structure distribution of the fished stock. Uncertainty in life-history parameters (triangular distribution) led to SPR estimates that were comparable to those derived from “well-known” values (four life-history parameters sets individually used). However, fishing mortality will be overestimated, and SPR underestimated if exploitation is focused on juveniles while older individuals are partially inaccessible to fisheries. In these cases, the LB-SPR outputs should be carefully interpreted and additional reference points, related to well-recorded sizes (e.g., juveniles), should be considered.

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2.3. Improving stock assessment and management advice for data-poor small-scale fisheries through participatory monitoring



**Improving stock assessment and management advice for data-poor small-scale fisheries through participatory monitoring**

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**Abstract**

Undetected but underlying biases in model parameterization strongly reduce the reliability and value of assessments of data-poor fisheries. We explore the effects of missing and misunderstood data on single-species stock assessments used to provide management advice. From 2006 to 2014, the Colombian government monitored landings of small-scale fisheries. During the same period, communities implemented a participatory monitoring program in the Central Guajira region. We found that the two data sources gave different results for the population status of the highest-valued fish, lane snapper (*Lutjanus synagris*), and the largest-landed species, white grunt (*Haemulon plumieri*). Recordings of landing points by the government monitoring program led to year-to-year underestimations and therefore misconceptions regarding population status and fishery trends. Overexploited and underexploited population statuses were seen to arise from the same fishing pressure as a result of the interplay between natural mortality and erroneous estimates of fishing mortality. The tested von Bertalanffy growth parameters affected the exploitation level, but not the population status, of the species. When data from the participatory monitoring program were incorporated, higher landings and a more severe overfishing trend emerged for both species. The management scenarios simulated using the best verified data available provided reasonable advice for recovering the lane snapper and white grunt populations. Furthermore, simulation of management measures sustained the employment and

incomes of fishers. Our findings indicate that participatory monitoring should be incorporated into the assessment and management of data-poor resources.

Keywords: Data-poor fisheries, stock assessment, government monitoring, participatory monitoring, lane snapper, white grunt

### **Introduction**

Most of the world's fish stocks are either lacking sufficient information or already depleted (Costello et al., 2012; Worm and Branch, 2012). This situation is critical because a large proportion of fisheries are located in developing regions, where increased demand for food and employment coincides with a weak control of fish stocks (Salas et al., 2007). Therefore, suitable integration and interpretation of available data on fisheries and stock status are key challenges. Both the quantity and quality of data on effort, landings and catch composition are important for assessing data-poor resources (Chen et al., 2003; Omori et al., 2016). Having fewer, but more reliable, fishery-dependent data could allow limited, but more robust, management advice to be provided for some species. Conversely, misconceptions of the real scope of available data may lead to misleading estimates of the historical and current pressures on fisheries.

The assessment of fisheries involves dealing with several sources of uncertainty (Scott et al., 2016). These uncertainties are usually interwoven with the underlying bias of the fishery model assumptions for evaluating target species in data-poor conditions (for instance, steady state; Rätz et al., 2010). Performing sensitivity tests and using the “Robin Hood” approach, which borrows data from the most reliable and related stocks, are recommended for improving model parameterization (Carruthers et al., 2014; Kokkalis et al., 2015). However, data-poor fisheries are challenged by unreported catches (Salas et al., 2007; Zeller and Pauly, 2007), leading to unpredictable estimates of trends in stock size and fishing effort (Omori et al., 2016). Consequently, misconceptions of the definition of fishing mortality result in highly risky results (Thorson and Prager, 2011), usually matched by a biased estimate of the life-history parameters of the studied targets from fishery-dependent data. This bias stresses the importance of “correctly” estimating natural mortality (Clark, 1999; Kenchington, 2013). Therefore, cumulative

bias from multiple sources can magnify the uncertainty associated with the single-species stock assessment of data-poor fisheries (Smith et al., 2009).

The limitations of data-poor assessments appear to be sufficiently identified but poorly addressed. This occurs because stock assessment involves parameterizing a model, but the inputs cannot be verified in a data-poor context. Frequently, unverified information is the starting point for trying to improve the available data. For instance, the reconstruction of landings does not usually consider the impact of fishing effort 1) among fleets, 2) on landing points, or 3) on changes in fishing regime (Lindop et al., 2015; Wielgus et al., 2010; Zeller and Pauly, 2007). Thus, reconstruction of landings is based on several assumptions whose validation is often impossible (Chen et al., 2003).

To address these issues, we take as a case study the Amerindian Wayuu fishery located in La Guajira in the Colombian Caribbean. While it is recognized that fishery landings in the Central Guajira occur at many points along the 90-km stretch of coastline (Manjarrés, 2004), from 2006 to 2014, the government monitoring program (GOV) and subsequent management advice focused exclusively on urban areas (CCI, 2006; Puentes et al., 2014). During the same period, the Wayuu people conducted a participatory monitoring program (PAR) to record catches at most landing points, including urban and rural ones.

We explored the reliability of advice derived from single-species stock assessment for this data-poor fishery. To this end, we studied the performance of stock assessment associated with the official data provided by the GOV as a single entity and combined with the PAR data. We distinguish between fisheries with unverified data, also known as "data-random" fisheries, which can lead to misleading advice, and "data-poor" fisheries, in which the available data are optimized (Bentley, 2015).

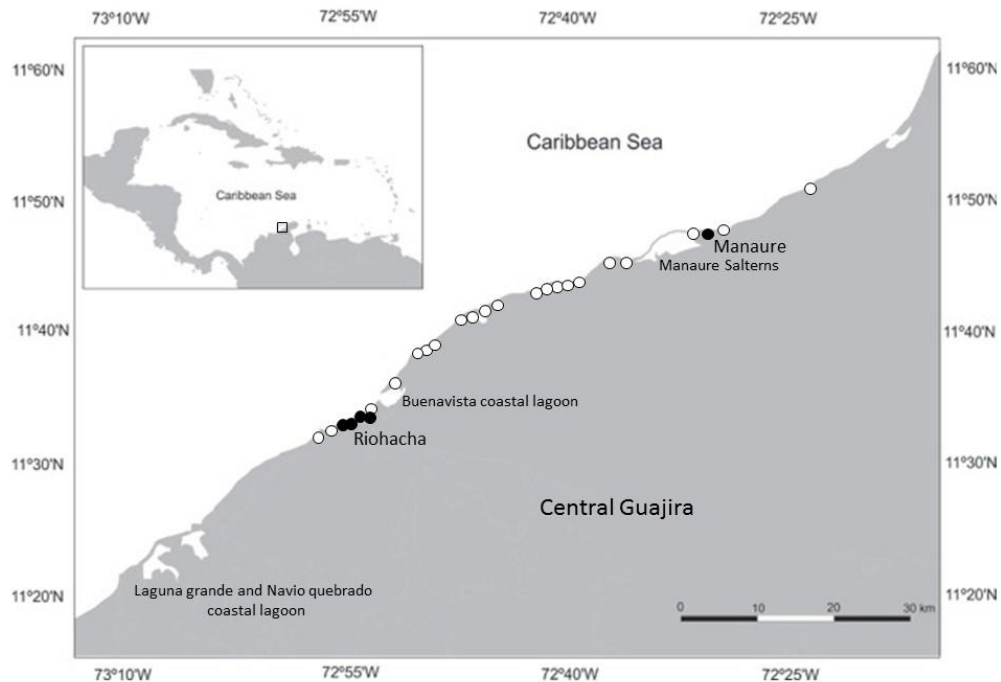


Fig. 16. Map of central Guajira, showing its position in the Caribbean Sea. Urban areas where the National fishery agency records the fishing landings are shown in filled circles, while rural areas where the Wayuu people performed the participatory monitoring are represented by empty circles.

The Wayuu fisheries are multi-species and multi-gear (Puentes et al., 2012). However, we selected the white grunt, *Haemulon plumierii*, the most important demersal species according to the landings, and the lane snapper, *Lutjanus synagris*, which fetches the highest market price among fish (CCI, 2006; Manjarrés, 2004), for this study. These two species are of great interest to Wayuu fishers, as the lane snapper covers income expectations and the white grunt meets social and food demands (J. Ramirez Pers. Comm.). We studied the effect of the GOV and PAR on the inputs of the length-based stock assessment models for both species (Leonart and Salat, 1997; Rätz et al., 2010). Here, we paid special attention to the impact of the reported landings, von Bertalanffy growth parameters ( $vBGP$ ), and natural mortality ( $M$ ) (Abella et al., 1997; Gislason et al., 2010; Kenchington, 2013). Stock assessments of the lane snapper and white grunt were performed using population analysis based on the interaction among gear over a limited time series. Finally, we analyzed several management scenarios aimed at achieving a sustainable fishery for both species in parallel. The harvesting sizes, fleet deployment, catch volume, food security and incomes of local communities were taken into account in the development of the management simulations.

### **Methods**

#### Study area

La Guajira is the northernmost region of South America (Fig. 16). Its central area has a wide and shallow continental shelf influenced by upwelling (Andrade and Barton, 2005; Sarmiento-Devia et al., 2013). These conditions promote high fishing activity (Manjarrés, 2004). Artisanal fisheries are dominant in the area, whereas trawl and larger long-line fleets are scarce (CCI, 2010; Zuñiga et al., 2004).

#### **Data collection**

The GOV database provided total landings by day from 2006 to 2011, but not the number of vessels. From November 2012 to July 2014, daily landings and the number of vessels were recorded. Throughout the study period, the catches of all vessels that arrived at five urban landing points in the Central Guajira were recorded (CCI, 2006). Data collection for the target species included the length frequency and macroscopic definition of maturity (CCI, 2006). For lane snapper, this information was only available in 2013. Little biological data for white grunt were collected because this species is not prioritized in the official fisheries advice (Barreto and Borda, 2008; Puentes et al., 2014).

A participatory monitoring program implemented by a local non-governmental organization (Fundación Ecosfera) and the Wayuu people was conducted during the same period. Through the participation of 19 communities, information from 85,347 daily trips (2006-2014) was recorded, based on over 18 rural and 3 urban landing points. The trained members of each community randomly collected landing data on four days a week. The total landings by species from the PAR were estimated using a daily raising factor - the number of sampled to unsampled active vessels. Additionally, the PAR collected length frequencies and maturity information for white grunt and lane snapper in 2010.

#### **Data treatment**

##### Landings reconstruction

Interpolation from reliable data was used to fill the gaps in landings for which the period and landing point were missing, following the methodology established by

the Sea Around Us Project (Zeller and Pauly, 2007). Three landing points in urban areas, sampled at the same time by both the government and participatory monitoring programs, were used to determine the degree to which landings in urban areas were underestimated. The total landed catch and selectivity pattern by gear of both species were calculated separately for the GOV and the PAR. Because daily landings by each vessel were not available for the GOV from 2006 to October 2012, the catch per unit effort (CPUE) was annually calculated from total landings and active vessels by fleet. Additionally, by combining the available length frequencies and landings by fleet, we were able to estimate the average size of the fish caught per year.

#### Life-history parameters

The *vBGP* were estimated using a non-linear model fit to the modal progression data with the Bhattacharya method, the selected asymptotic length ( $L_{\infty}$ ) and the growth constant rate ( $k$ ) according to the FISAT II routines (Gayanilo et al., 2005; Pinheiro et al., 2015). The *vBGP* of lane snapper were obtained from the encircling gillnets ( $n=1271$ ) of the GOV in 2013. Using data from the bottom gillnets ( $n=541$ ), long-lines ( $n=1055$ ) and encircling gillnets ( $n=99$ ) of the PAR, an additional set of *vBGP* were calculated. In the case of the white grunt, length frequencies were only available from the PAR study. Here, bottom gillnets ( $n=977$ ), long-lines ( $n=181$ ) and encircling gillnets ( $n=15$ ) provided the data to estimate the *vBGP*.

The *vBGP* calculated from fishery-dependent data on a population subjected to strong exploitation are probably biased (Then et al., 2015). We therefore explored several growth parameters for lane snapper derived from the literature (Acosta and Appeldoorn, 1992; Barreto and Borda, 2008; Frédou et al., 2009; Gómez et al., 2001; Johnson et al., 1995; Leite Jr. et al., 2005; Luckhurst et al., 2000; Manickchand-Dass, 1987; Manjarrés, 2004; Puentes et al., 2014). Only the *vBGP* obtained in Trinidad and Tobago by Manickchand-Dass (1987) for females met the conditions of 1) being based on otolith readings, 2) showing an asymptotic length higher than the maximum length found in the Central Guajira, 3) including small and large sizes and showing environmental conditions similar to our study area, and 4) being insensitive to changes in the value of terminal fishing mortality (explained below). However, both the  $L_{\infty}=60$  cm and  $k=0.20 \text{ yr}^{-1}$  obtained from this study were similar to our own ( $L_{\infty}=64.94$  cm,

$k=0.19 \text{ yr}^{-1}$ ). To explore a wider range of plausible states of nature, the sensitivity analysis further considered  $L_{\infty}=63.75 \text{ cm}$  and  $k=0.27 \text{ yr}^{-1}$ , which were previously used in Colombian assessments (see Puentes et al., 2014). Using the above criteria for lane snapper, a second set of  $vBGP$  for white grunt was selected from the available literature (Araújo and Martins, 2007; Dominguez-Viveros, 1996; Murie and Parkyn, 2005; Potts and Manooch, 2001; Valle et al., 1997). None of the  $vBGP$  met all the criteria. Therefore,  $L_{\infty}=59.10 \text{ cm}$  and  $k=0.08 \text{ yr}^{-1}$  (Potts and Manooch, 2001) were explored, though they belong to a population that inhabits colder water.

Given its high uncertainty (Kenchington, 2013; Scott et al., 2016; Then et al., 2015),  $M$  for both species was calculated using several indirect estimators. Generic  $M$  estimators based on life-history correlates, such as Pauly's estimator and Ralston's estimator (focusing on lutjanids) were explored (Kenchington, 2013). Hoenig's estimator based on maximum estimated ages ( $T_{\max}$ ) was used (Newman, 2002; Newman et al., 2000), as were estimators of  $M$ -at-age (Abella et al., 1997) and  $M$ -at-size (Gislason et al., 2010).

$M$  estimators based on  $T_{\max}$  were applied based on the oldest ages from the literature. The ages were considered valid when they came from populations that were either lightly exploited or not overexploited. The oldest selected age for lane snapper was 19 years (Luckhurst et al., 2000). Additionally, a natural mortality of  $0.13 \text{ yr}^{-1}$ , estimated by the catch-curve method for unfished populations of other lutjanids, was included in the analyses (Newman, 2002; Newman et al., 2000). Considering that large differences in the oldest age of white grunt have been found (see Araújo and Martins, 2007; Murie and Parkyn, 2005), we considered maximum ages of 18 and 28 years for the species.

The data used to calculate female maturity for lane snapper ( $n=504$ ) and white grunt ( $n=660$ ) came from the PAR data set. Individuals were considered mature as of stage II (developing) by macroscopic observation because lane snapper and white grunt can begin to spawn when they have reached the developing stage (Freitas et al., 2014; ICES, 2008; Shinozaki-Mendes et al., 2013). Different analytical procedures for obtaining the maturity-at-length curve were explored (ICES, 2008). The logistic function is usually used for fitting the proportion of mature fish by size (or age) and obtaining parameters,

particularly L50. The cumulative normal distribution fits, as well as the logistic function, has the advantage of being more theoretically justifiable, as maturation occurs following a probability law. However, both models are theoretically problematic, as negative values of the independent variable, which are not defined, yield positive values of maturation, which is nonsensical. Thus, it is more justified to fit a lognormal cumulative curve that meets all the theoretical requirements and explains the maturation process equally well or better. The cumulative lognormal gave the best fit to the data with the following maturity-at length parameters for the lane snapper (L25= 18.94 cm, L50= 22.77 cm, L75= 29.61 cm) and the white grunt (L25= 15.39 cm, L50= 16.33 cm, L75= 19.21 cm). Weight-at-length relationships were calculated from the PAR data using  $W = aL^b$  (where  $a = 0.0173$  and  $b = 2.9168$  and  $a = 0.008$  and  $b = 3.1715$  for the lane snapper and white grunt, respectively). Given limited data for calculating maturity-at-length, the sensitivity tests performed in this study do not account for the uncertainty associated with maturity ogive calculation.

#### Sensitivity tests and population dynamics

We assumed that the size structure of species caught with the three types of fishing gear did not change during the study period because length-frequency data were only available for the bottom gillnet and long-line in 2010 and for the encircling gillnet in 2013. The stock assessment was performed using the VIT software, which was specially developed to conduct stock assessment when catch-at-length data are limited in duration (Leonart and Salat, 1997). Virtual population analysis (VPA), yield per recruit (Y/R) and management simulations were performed using catch data structured by age (Aldebert and Recasens, 1996; Rätz et al., 2010). Furthermore, length data, maturity-at-length and mortality-at-length were converted to age data using a slicing method based on the von Bertalanffy growth equation. The biological process could thus be propagated through age-based data.

Since the age groups of a given year are used to obtain a single cohort from pseudo-cohort assumptions, one VPA was performed for each individual year (Leonart and Salat, 1997; Rätz et al., 2010). The terminal fishing mortality ( $F_t$ ) is required to start the VPA for each individual year, so we performed a sensitivity analysis of this parameter (0.05, 0.1, 0.2, 0.3, 0.4, 0.5, 1.0 and 1.5  $\text{yr}^{-1}$ ) on the VPA results, keeping the same  $vBGP$

and  $M$ . Those  $vBGP$  that led to the same fishing mortality trend among ages when  $F_t$  changed were chosen, and the others were rejected. Terminal fishing mortalities from 0.05 to 0.5  $\text{yr}^{-1}$  led to similar stock results, so an intermediate value of  $F_t=0.2 \text{ yr}^{-1}$  was selected for both species. The plus group ranged from two to four for lane snapper and from five to six for white grunt, depending on the  $vBGP$  used.

According to the selected  $vBGP$  for each species and natural mortality estimated by different indirect estimators, as mentioned above, we conducted sensitivity analyses to explore the parameters' impact on assessment outputs. For these analyses, we used the data from 2006 as the first year with available data of landings and length structure by fleet. From the three sets of  $vBGP$  and six estimates of natural mortality, the sensitivity analyses of lane snapper included 18 VPA and Y/R. In the case of the white grunt, we used three  $vBGP$  and five different estimates of natural mortality, producing 15 VPA and Y/R. During the sensitivity analyses, only the  $vBGP$  and  $M$  were modified, keeping the other inputs fixed. The sensitivity results were plotted using an adapted Kobe plot (Nishida et al., 2014). Virgin biomass estimators using VIT are uncertain and not very reliable (Rätz et al., 2010). However, these authors found that the overall trends of fishing mortality and stock size by VIT are well captured in comparison to a traditional VPA. To make the stock assessment comparable among years, we selected, as a relative measure of exploitation, the ratio between current fishing mortality and that which produces a maximum sustainable yield ( $F/F_{msy}$ ). For the abovementioned reason, we consider the  $F_{max}$  and its yield as proxies for  $F_{msy}$  and MSY. Additionally, we considered two relative reference points according to the maximum (virgin) spawning stock biomass estimated in each year ( $SSB/SSB_{max}$ ):  $SSB_{40\%}$ , as a precautionary reference point when stock recruitment relationships were unknown (Myers et al., 1994), and  $SSB_{20\%}$ , defined as the fishing level that reduces spawning per recruit to a threshold biomass level of 20% of the unfished stock (Rosenberg, 1993).

#### Management strategy simulations

The initial age-structure at year 0 to start the management simulations was defined according to the average of the last two years of the reconstructed data (2013 and 2014). We took into account control of the fleet effort (which, in the model, acts as an  $F$  multiplier,  $\phi$ ) in terms of vessel numbers, incomes, and food security for the Central

Guajira communities. Considering it mandatory to reach a relative SSB20% within 8 years, the current  $F/F_{msy}$  was adjusted by changing the fishing multiplier by gear. We explored several scenarios for managing the lane snapper and white grunt (Table 9). A change in fishing effort by gear could be not required ( $\phi=1$ ), increased ( $\phi>1$ ), reduced ( $\phi<1$ ), or gear eliminated ( $\phi=0$ ). The management simulations also studied the effect of eliminating the effort of two fleets and allowing the optimization of the third fleet effort to achieve MSY. For instance, the “people first” scenario prioritizes gillnets because it is the gear most used by rural fishers and it catches medium-sized fish. In this case, the efforts of long-line and encircling gillnet fleets are reduced to zero, whereas the fishing effort of gillnets is increased as much as possible.

Simulation under different life-history and stock-recruitment models extended the sensitivity analyses. Two lane snapper  $vBGP$  sets were selected:  $L_{\infty}=64.94$  cm,  $k=0.19$  yr<sup>-1</sup>,  $t_0=-1.58$  yr and  $L_{\infty}=63.75$  cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-0.66$  yr. The first were calculated from the widest range of sizes available from the PAR and the second were those used by the government assessments. However, with white grunt, only a single  $vBGP$  set from the Central Guajira was available ( $L_{\infty}=41.65$  cm,  $k=0.26$  yr<sup>-1</sup>,  $t_0=-1.43$  yr), but a wider range of simulations was conducted based on the two calculated natural mortality rates (0.15 yr<sup>-1</sup> and 0.24 yr<sup>-1</sup>). The Beverton and Holt model (B&H) (Beverton and Holt, 1959) was used to simulate recruitment ( $R=1/(\alpha+SSB/\alpha)$ ). This model was used to simulate a constant recruitment ( $\alpha=1$ ,  $\beta\rightarrow 0$ ) or a quasi-linear one ( $\alpha=0.5$ ,  $\beta=0.5*SSB$ ). The constant recruitment was initially used because recruitments and reliable SSB by year were unknown. Additionally, a quasi-linear model was used by assuming that the management measures could be adequately implemented by stakeholders from year one and that pre-recruitment phases of target species would not be affected by the recruitment phase. The number of stochastic iterations of the lognormally distributed stock-recruitment model for each run was set at 1000 with a mean of 1 and a variance of 0.1. The yield per recruit was represented by the mean with 95% confidence intervals.

## Results

### Catch reconstruction

From 2006 to 2014, the total reconstructed catch of lane snapper was estimated at 410 tons and the total catch of white grunt at 2,017 tons. The GOV data underestimated landings. The reconstructed landings of lane snapper and white grunt ranged from 3.25 to 8.52 times and 1.38 to 3.70 times greater than the official GOV landings (Fig. 17). Most landings of lane snapper occurred in rural areas unmonitored by the GOV. In the case of white grunt, the official and reconstructed catches showed similar trends from 2006 to 2009. As of 2010, higher landings in rural areas led to the greater underestimations of government statistics (Fig. 17b). The GOV data suggested that the encircling gillnet fleet took the largest proportion of lane snapper (mean, 83%) and white grunt (mean, 99%). The reconstructed landings of lane snapper showed that the long-line fleet took the bulk of the catch during the first few years but that the catches have shifted to being dominated by encircling and bottom gillnets since 2011 (Fig. 18a,b). The reconstructed landings of white grunt increased the participation of bottom gillnets to 20% (Fig. 18c, d). On the other hand, the reduction of the average size of lane snapper remained hidden in the official GOV landings, and the reconstructed landings show the size to have decreased during the study period (Fig. 18). The average size of white grunt among the years remained stable in both monitoring programs.

<i>Lutjanus synagris</i> management scenarios		Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher and high-valued catches and ensuring food security in the rural communities (long-line and bottom gillnet) Optimizing both highly valued sizes (long-line) and
2	The "best" catches	Removing gear that fishes the highest proportion of medium sizes (bottom gillnet)	Eliminating the fleet formed by the most vessels from rural communities (bottom gillnet)	the highest catches (encircling gillnet), even if food security in rural areas could be affected (bottom gillnet)
3	Only harvesting medium sizes	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and small sizes (encircling gillnet)	Keeping the fleet formed by the most vessels from rural communities (bottom gillnet)	Encouraging both the cheapest fishing trips and food security, especially in rural areas
4	The volume does matter	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and medium sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
5	Targeting the highest-valued sizes	Reducing as much as possible catches with a high proportion of both small sizes (encircling gillnet) and medium sizes (bottom gillnet)	Keeping the fleet that underwent the greatest fishing reduction during the study period	Promoting the highest-valued as well as the lowest catches. Food security of rural communities is poorly considered
6	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)
<i>Haemulon plumieri</i>		Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher catches and ensuring food security in the rural communities (bottom gillnet)
2	The volume does matter	Removing gear that fishes the highest proportion of medium and large sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
3	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)

Table 9. Management scenarios of lane snapper and white grunt fisheries according to fish size, deployed fleet and food security and incomes of fishing communities. Management scenarios are separated by species given that the fleets act differently on sizes and total catches. Management scenarios were incorporated into simulations by controlling the caught sizes by gear (Fish size column) and fleets in the fishery (Deployed fleet column). The third column (Food security and profits of fishing communities) shows the effect caused by controlling for fish size and fleet, but it is not directly incorporated into the simulation model.

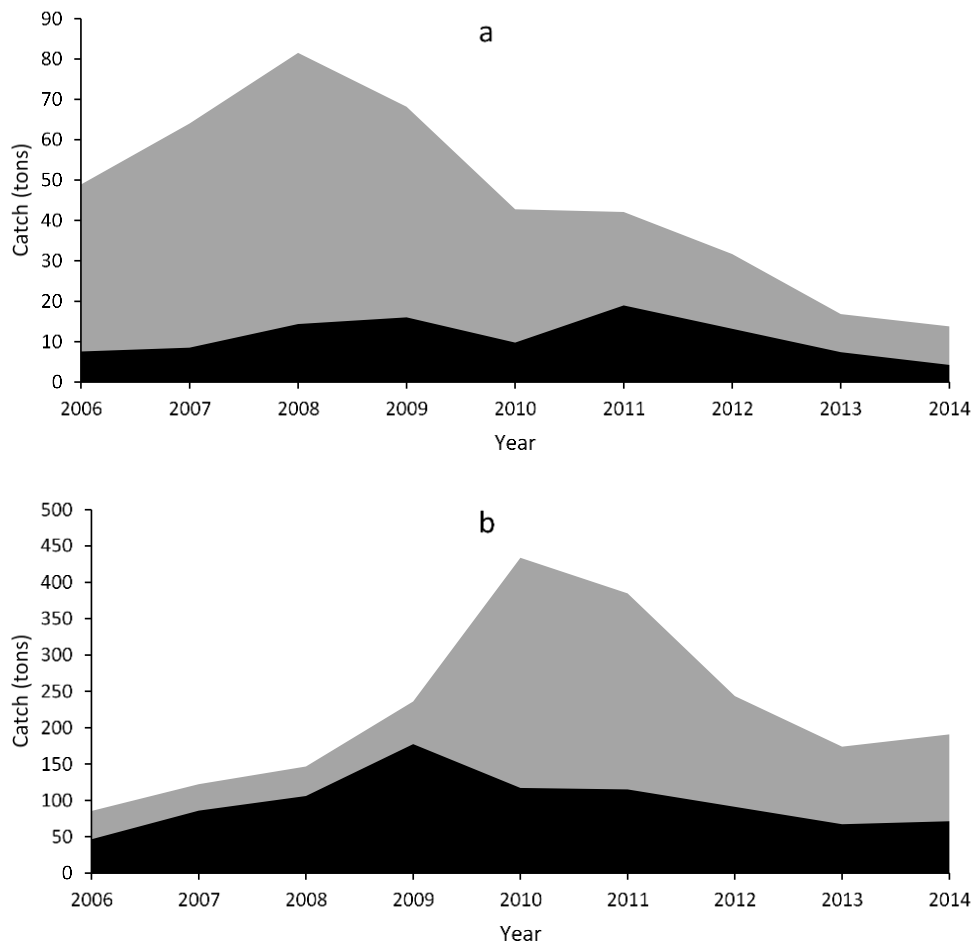


Fig. 17. Reported catch of lane snapper (a) and white grunt (b) by the National fishery agency involving urban areas (black), and total reconstructed catch (gray) employing both the rural landings by participatory-monitoring program and the urban catches by Government-monitoring program.

### Life-history parameters

The catches of lane snapper from the bottom gillnet and long-line fleets (from PAR) were mainly composed of sizes ranging from 14 to 40 and from 20 to 50 cm, respectively (see supplementary material, table S1). The encircling gillnet fleet (from GOV) mostly included sizes ranging from 19 to 41 cm (see supplementary material, table S2). Consequently, the  $vBGP$  calculated from the long-lines and bottom gillnets ( $L_{\infty}=64.94$  cm,  $k=0.19$  yr<sup>-1</sup>,  $t_0=-1.58$  yr) produced a higher  $k$  than the  $vBGP$  calculated from the encircling gillnets ( $L_{\infty}=62.5$  cm,  $k=0.15$  yr<sup>-1</sup>,  $t_0=-3.19$  yr). The length-based analysis for the white grunt population mainly included data from bottom gillnets (see supplementary material, table S3). Thus, the white grunt ranged from 15 to 36 cm, with a high  $k$  value ( $L_{\infty}=41.65$  cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-1.43$  yr).

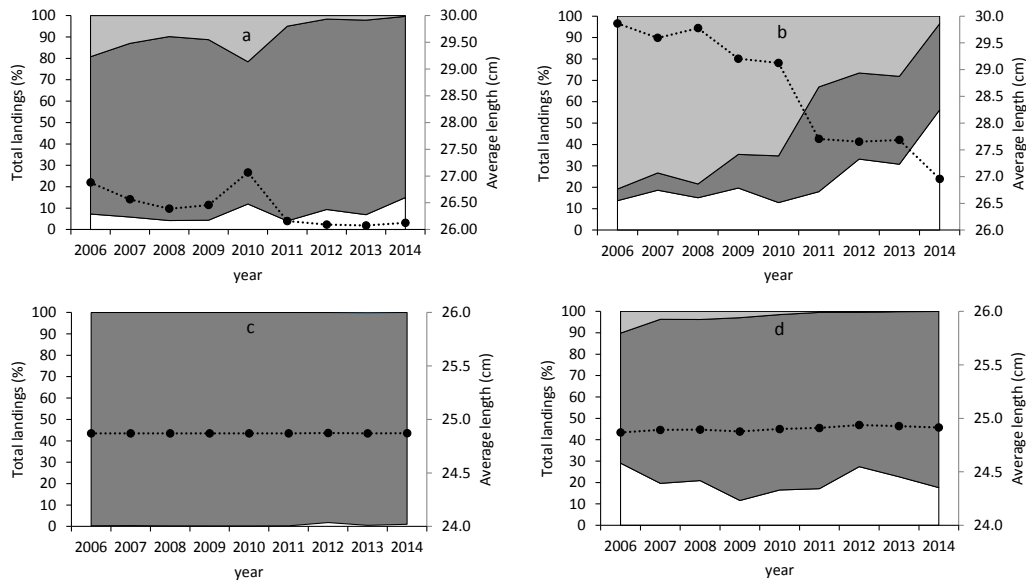


Fig. 18. Percentage of participation of longline (light grey), bottom gillnets (white) and encircling gillnets (dark grey) in the catches of lane snapper (a and b) and white grunt (c and d) from 2006 to 2014 on the left y axis. Average size of caught individuals on the right y axis (dotted line). Government-monitoring program (a and c) and Government-participatory combined data (b and d).

Natural mortality estimators based on life-history correlations tended to produce higher natural mortality rates in both species (Pauly's  $M=0.41-0.67 \text{ yr}^{-1}$  and Ralston's  $M=0.30-0.42 \text{ yr}^{-1}$ ). Similar results were produced by Gislason's *M-at-size* estimator (lane snapper average size,  $M=1.21-0.62 \text{ yr}^{-1}$ ; white grunt average size,  $M=1.97-0.31 \text{ yr}^{-1}$ ; see supplementary material, table S4). Additionally, the estimates from *M* estimators varied substantially depending on the  $L_{\infty}$  and  $k$  values used. Conversely, Hoenig's estimator based on  $T_{\text{max}}$  produced lower or moderate natural mortality rates (lane snapper,  $M=0.23 \text{ yr}^{-1}$  for  $t_{\text{max}}=19 \text{ yr}$ ; white grunt,  $M=0.15 \text{ yr}^{-1}$  for  $t_{\text{max}}=8 \text{ yr}$  and  $M=0.28 \text{ yr}^{-1}$  for  $t_{\text{max}}=18 \text{ yr}$ ). Abella's *M-at-age* estimator, which is also based on life-history correlations, produced lower mortality rates. The average of the *M* values for the first six ages ranged from 0.10 to 0.18  $\text{yr}^{-1}$  for lane snapper and from 0.12 to 0.20  $\text{yr}^{-1}$  for white grunt (see supplementary material table S5).

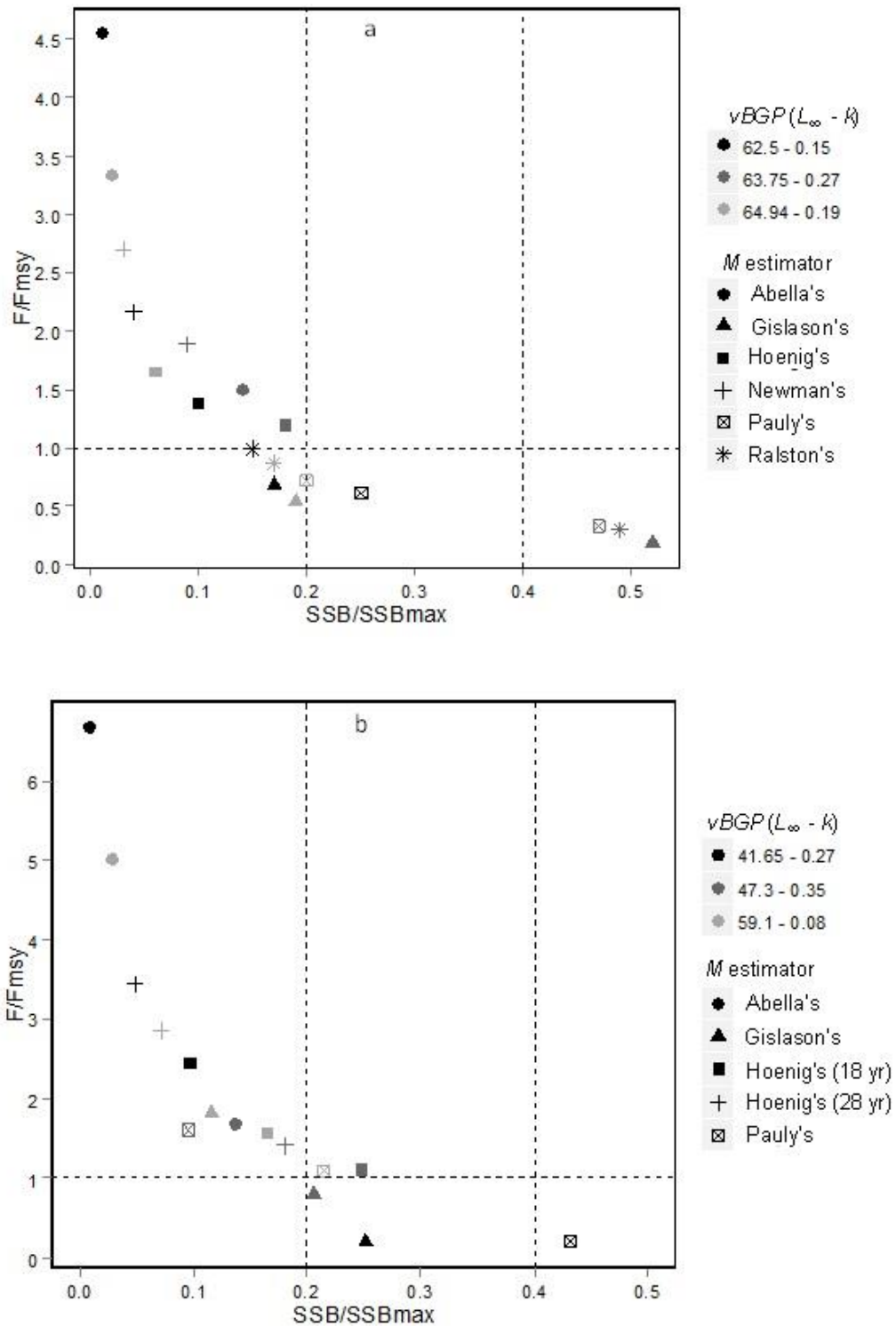


Fig. 19. Modified Kobe plot, involving three pairs of  $vBGP$  (symbols sharing the same color) and different natural mortality ( $M$ ) estimators (Different symbols). Life-history correlates for constant  $M$  such as the Pauly's (square and equis), Ralston's estimators (asterisk); the Gislason's  $M$ -at-size estimator (triangle); Abella's  $M$ -at-size estimator (circle);  $M$  based on maximum age ( $T_{max}$ ) by Hoening's (square) and  $M=0.13$  based on catch-curves from some lutjanids (plus). Hoening's estimator was performed to 19 years (lane snapper) and 18 and 28 year (white grunt) of maximum estimated age. Ralston's estimators were performed just for lane snapper, since these estimators are specific for lutjanids. Additional value from Hoening' estimator to white grunt is showed with plus symbol. Y/R analysis performed for populations of lane snapper (a) and white grunt (b) at the Central Guajira.

### Sensitivity tests

The population status of both species ranged from highly overexploited to underexploited (Fig. 19). This was mainly due to the natural mortality assumed in each stock assessment. The natural mortality estimated from Pauly's, Ralston's and Gislason's estimators yielded no overfished stocks ( $SSB/SSB_{max} > 40\%$ ) and overfished stocks ( $SSB/SSB_{max} < 20\%$ ), but overfishing was not occurring ( $F/F_{msy} < 1$ ). Conversely, stock assessments based on the  $M$  from Hoenig's and Abella's estimators indicated overexploitation. The exploitation levels of lane snapper and white grunt were also largely influenced by changes of  $vBGP$ . Abella's estimator showed the largest changes in  $F/F_{msy}$ . Gislason's, Pauly's and Ralston's estimators led to a large variation in  $SSB/SSB_{max}$ . Hoenig's estimator, which is based on the maximum estimated age, led to the lowest variation in exploitation level (Fig. 19).

### Population status

We chose three and two sets of  $vBGP$  to describe the growth of the lane snapper and white grunt, respectively. In the case of the lane snapper, the landings reduction and increased fishing mortality of small-sized fish during the study period suggested overfishing. Additionally, the high values of natural mortality contravene the low to moderate estimates of  $M$  obtained from unexploited snapper populations using catch-curve methods (Newman, 2002; Newman et al., 2000). For lane snapper, Frédou et al. (2009) suggested low  $M$ . In performing a stock assessment of the lane snapper, we selected natural mortality rates of 0.13 and 0.24  $yr^{-1}$  from the Newman et al. (2000) and Hoenig's estimators. Further complications emerge for selecting  $M$  for the white grunt, as larger gaps in its estimation remain when indirect estimators are used. Therefore, we selected the natural mortality rates of 0.15 and 0.24  $yr^{-1}$  by focusing on methods that use the maximum estimated age, as suggested by Then et al. (2015), and avoided using  $M$  values higher than 0.35  $yr^{-1}$ , as recommended by Potts (2000).

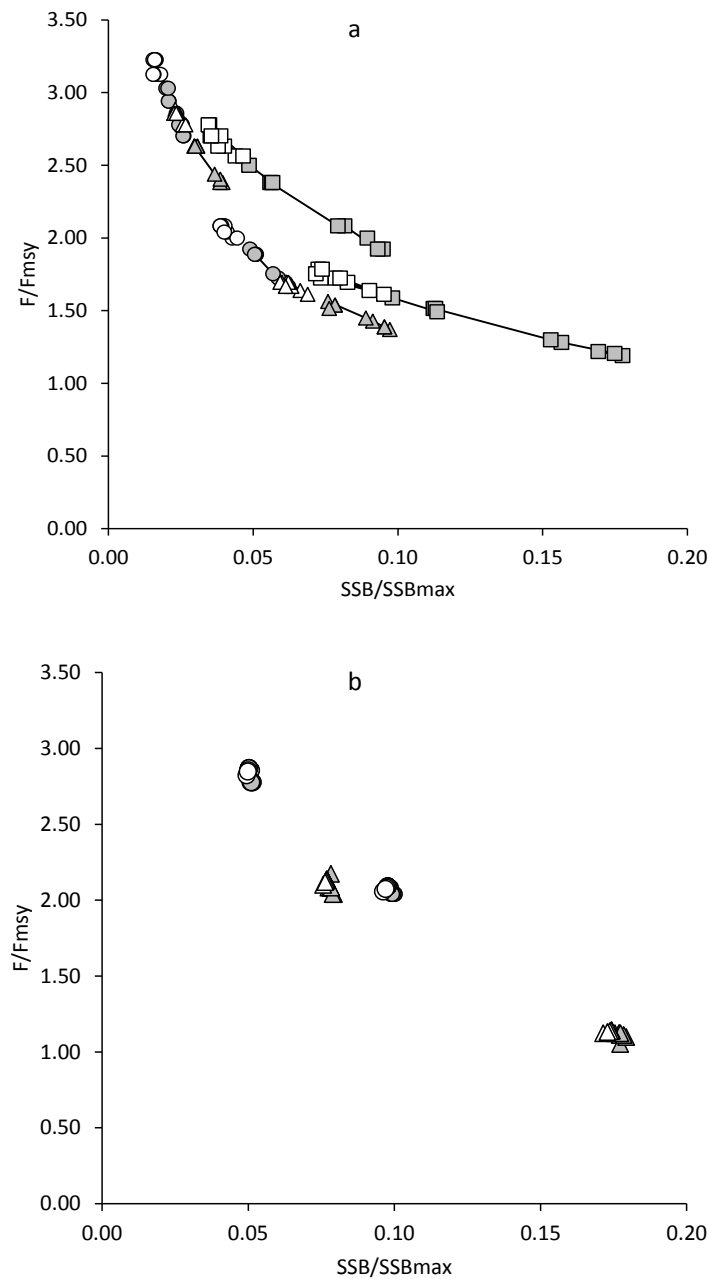


Fig. 20. Precautionary plot, establishing the relative threshold reference point ( $SSB_{20\%}$ ) and  $F$  ratio ( $F/F_{msy}$ ). Historic analysis performed from Government (empty symbols) and reconstructed landings (gray symbols). Late years show higher overexploited level. Lane snapper (a) when  $L_{\infty}=64.94$ ,  $k=0.19$ ,  $M=0.23$  (lowermost circles) and  $M=0.13$  (upper circles),  $L_{\infty}=63.75$ ,  $k=0.27$ ,  $M=0.23$  (lowermost squares) and  $M=0.13$  (upper squares) and  $L_{\infty}=63.75$ ,  $k=0.27$ ,  $M=0.23$  (lowermost triangles) and  $M=0.13$  (upper triangles). White grunt (b) when  $L_{\infty}=41.65$ ,  $k=0.26$ ,  $M=0.24$  (lowermost circles) and  $M=0.15$  (upper circles) and  $L_{\infty}=59.01$ ,  $k=0.08$ ,  $M=0.24$  (lowermost triangles) and  $M=0.15$  (upper triangles).

The lane snapper was estimated to be more overexploited when the assessment was based on landings from the GOV (Fig. 20a). Underestimation of long-line catches in urban areas meant that large fish sizes were unreported. Furthermore, the stock

assessment performed using the GOV data led to less change between earlier and later years than that obtained when the PAR data were also included in the assessment. This means that a worsening in the overexploitation status of the lane snapper was smoothed because the GOV did not perceive the increase in fishing mortality of smaller-sized fish.

The status of the white grunt population was insensitive to whether government and unified government-participatory monitoring data were analyzed (Fig. 20b). The encircling gillnet fleet showed a large participation in urban and rural landing points, reporting between 61% and 83% of the total reconstructed landings. Additionally, the encircling gillnet fleet caught fish sizes similar to the those for the bottom gillnet fleet, implying that between 89% and 99% of landings involved a similar length structure. These conditions contributed to an unchanged population status between the GOV and reconstructed data. Therefore, the total biomass harvested by the fleet should be a key aspect to consider in the management of the white grunt population.

#### Management simulations

The lane snapper and white grunt populations are estimated to have been overexploited since 2006. The dominance of the encircling gillnet fleet in lane snapper and white grunt catches, which mainly removed small-sized fish, placed the relative spawning stock biomass (SSB/SSB<sub>max</sub>) of both species below 20%. The trends suggest higher overfishing between the earlier and later years studied (Fig. 20). The simulation outputs covered the selected  $vBGP$  and  $M$ , as well as the two different recruitment models (Constant and B&H). The B&H recruitment model allowed less reduction of  $\bar{E}$  than the constant model for both species, regardless of which set of  $vBGP$  was used. The management scenarios “people first” ( $1 \geq \phi > 0.82$ ) and “only harvesting of medium sizes” ( $\phi = 1$ ) led to the lowest reduction of the bottom gillnet fleet effort (Table 10). From a social point of view, this is desirable because  $Y/R$  increased over initial simulation levels (see scenarios 1 and 3, Fig. 21). Furthermore, many fishers who inhabit rural areas would be able to secure income and food (Table 9). Conversely, if priority is given to obtaining the highest of either income or catch, the encircling gillnet fleet should be maintained (Table 9). This would, however, lead to a reduction in the effort of most bottom gillnet vessels (“the best catches”;  $0.28 \geq \phi \geq 0$ ) and long-line vessels (“the volume does matter”;

$0.47 \geq \phi \geq 0.2$ ). This means that rural communities would be seriously affected (Table 9 and Table 10), and the recovery of the lane snapper population would be both lower and slower than in previous scenarios (Fig. 21). Focusing the fishing effort on the highly valued large fish sizes increases long-line yields. This would require an important reduction in the effort of the bottom and encircling gillnet fleets ( $0.63 \geq \phi \geq 0.40$ ), which would be socially unsustainable.

The white grunt fishery has a marginal long-line harvest (lower than 5% of total catch), so it was not necessary to consider long-lines in the management scenarios. The “people first” scenario, which prioritized the bottom gillnet fleet, also allowed some participation of the encircling gillnet fleet ( $0.46 \geq \phi \geq 0.06$ ). This would prioritize the protection of employment and food security for rural and urban fishers (scenarios 1 and 2; Figure 7). Conversely, prioritizing the largest catches demands a reduction in the effort of the encircling gillnet fleet ( $0.76 \geq \phi \geq 0.4$ ) and the total elimination of the bottom gillnet fleet ( $\phi = 0$ ), a total reduction of 66.3% to 86.5% of the operative fleet effort (Table 10). Finally, a homogeneous reduction of all fleet efforts for both species would not offer higher yields but would result in greater negative impacts on fishery incomes and food security (Fig. 21 and 7).

When scenarios that favored the traditional bottom gillnet fleet (“people first” and “just harvesting of medium sizes”) were optimized, the fishing effort on lane snapper ( $1.62 \geq \phi \geq 1.1$ ) and white grunt ( $2.6 \geq \phi \geq 1.19$ ) could be increased. Optimization of a more dominant fleet, the encircling gillnets (“the volume does matter”), allowed the fishing effort on lane snapper to be increased ( $1.75 \geq \phi \geq 1.27$ ). However, this optimization would require harvesting of white grunt to be reduced to achieve SSB20% ( $\phi = 0$ ) (Table 10), suggesting that joint management of both species depends on the fishing effort of the encircling gillnets on the white grunt population.

Species	Scenario	vBGP and M	Recruit model	Management measures ( $\phi$ by gear)			Optimization of $\phi$
				Bottom gillnet	Encircling gillnet	Long-line	
<i>L. synagris</i> (Lane snapper)	People first	a	Constant	0.82	0	0.82	N/A
			B&H	1	0.15	1	1.1
		b	Constant	0.9	0	0.9	N/A
			B&H	1	0.27	1	1.2
	The "best" catches	a	Constant	0	0.92	0.92	N/A
			B&H	0.25	1	1	1.2
		b	Constant	0	0.9	0.9	N/A
			B&H	0.28	1	1	1.2
	Only harvesting medium sizes	a	Constant	1	0.12	0.12	1.15
			B&H	1	0.4	0.4	1.5
		b	Constant	1	0.16	0.16	1.2
			B&H	1	0.45	0.45	1.62
	The volume does matter	a	Constant	0.21	1	0.21	1.35
			B&H	0.45	1	0.45	1.75
		b	Constant	0.2	1	0.2	1.27
			B&H	0.47	1	0.47	1.65
	Targeting the highest-valued sizes	a	Constant	0.4	0.4	1	2.9
			B&H	0.6	0.6	1	3.9
		b	Constant	0.42	0.42	1	3.35
			B&H	0.63	0.63	1	4.7
	Lower fishing	a	Constant	0.51	0.51	0.51	N/A
			B&H	0.65	0.65	0.65	N/A
		b	Constant	0.52	0.52	0.52	N/A
			B&H	0.69	0.69	0.69	N/A
<i>H. plumierii</i> (white grunt)	People first	c	Constant	1	0.06	1	1.19
			B&H	1	0.22	1	1.7
		d	Constant	1	0.29	1	1.9
			B&H	1	0.46	1	2.6
	The volume does matter	c	Constant	0	0.4	1	N/A
			B&H	0	0.55	1	N/A
		d	Constant	0	0.6	1	N/A
			B&H	0	0.76	1	N/A
	Only harvesting medium sizes	c	Constant	0.3	0.3	1	N/A
			B&H	0.4	0.4	1	N/A
		d	Constant	0.46	0.46	1	N/A
			B&H	0.59	0.59	1	N/A

Table 10. Simulations of management scenarios, based on.  $L_{\infty}=64.94$  cm,  $k=0.19$  yr<sup>-1</sup>,  $t_0=-1.58$ ,  $M=0.23$  yr<sup>-1</sup> (a) and  $L_{\infty}=63.75$ ,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-0.66$ yr,  $M=0.13$  yr<sup>-1</sup> (b) to lane snapper and  $L_{\infty}=41.65$ cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-1.43$ yr,  $M= 0.15$  yr<sup>-1</sup> (c) and  $L_{\infty}=41.65$ cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0 =-1.43$ yr,  $M=0.24$  yr<sup>-1</sup> (d) for white grunt populations. Each scenario produced four results according to vBGP-M and either the constant or Beverton and Holt (B&H) recruitment models. Simulation cases in which the prioritized fleet could maintain a fishing multiplier ( $\phi$ ) above 1 were maximized, and other gears were eliminated (optimization column).

## Discussion

### Addressing reliable biomass removal by fishery

The Wayuu fishery in the Central Guajira of Colombia brought to light a large underestimation obtained from the official catch statistics. The reconstructed landings tended to decrease, but this behavior was not detected by the GOV. This is because the GOV only considered major urban landing points, whereas fishing vessels are also landing on a wide rural shoreline, as occurs in many developing countries (Salas et al., 2007). Inter-annual changes in the landing areas according to fleet also affected the reported catches. The long-line fleet (which decreased during the study period) and the bottom gillnet fleet mainly landed in rural areas. The encircling gillnet fleet landed in urban areas prior to 2010 but shifted to landing in both urban and rural areas thereafter. Consequently, the GOV recorded only a portion of the entire harvest for each year. Such discrepancies can lead to an unpredictable bias in the estimation and trend of stock size, as already noted by Omori et al. (2016).

Our study shows evidence for a major change to the most used gear in the lane snapper fishery. Additionally, at least three different regimes of fishing effort have been identified since 1982 in the Colombian Caribbean (García, 2010; Manjarrés, 2004). Accordingly, we used several raising factors to separate the effects of port, fleet and period (i.e., month and year) on the reconstructed landings. To date, the reconstructions of marine landings in Colombia and other developing countries have used a single raising factor for all time periods (Lindop et al., 2015; Wielgus et al., 2010; Zeller and Pauly, 2007), thus ignoring differences in harvesting power among fleets and the effects of different fishing effort regimes on the reported landings. The reconstruction of world landings highlights the importance of small-scale fisheries for attaining resource sustainability and for providing food security. However, this reconstruction could lead to an incorrect estimate of fishing pressure and population status.

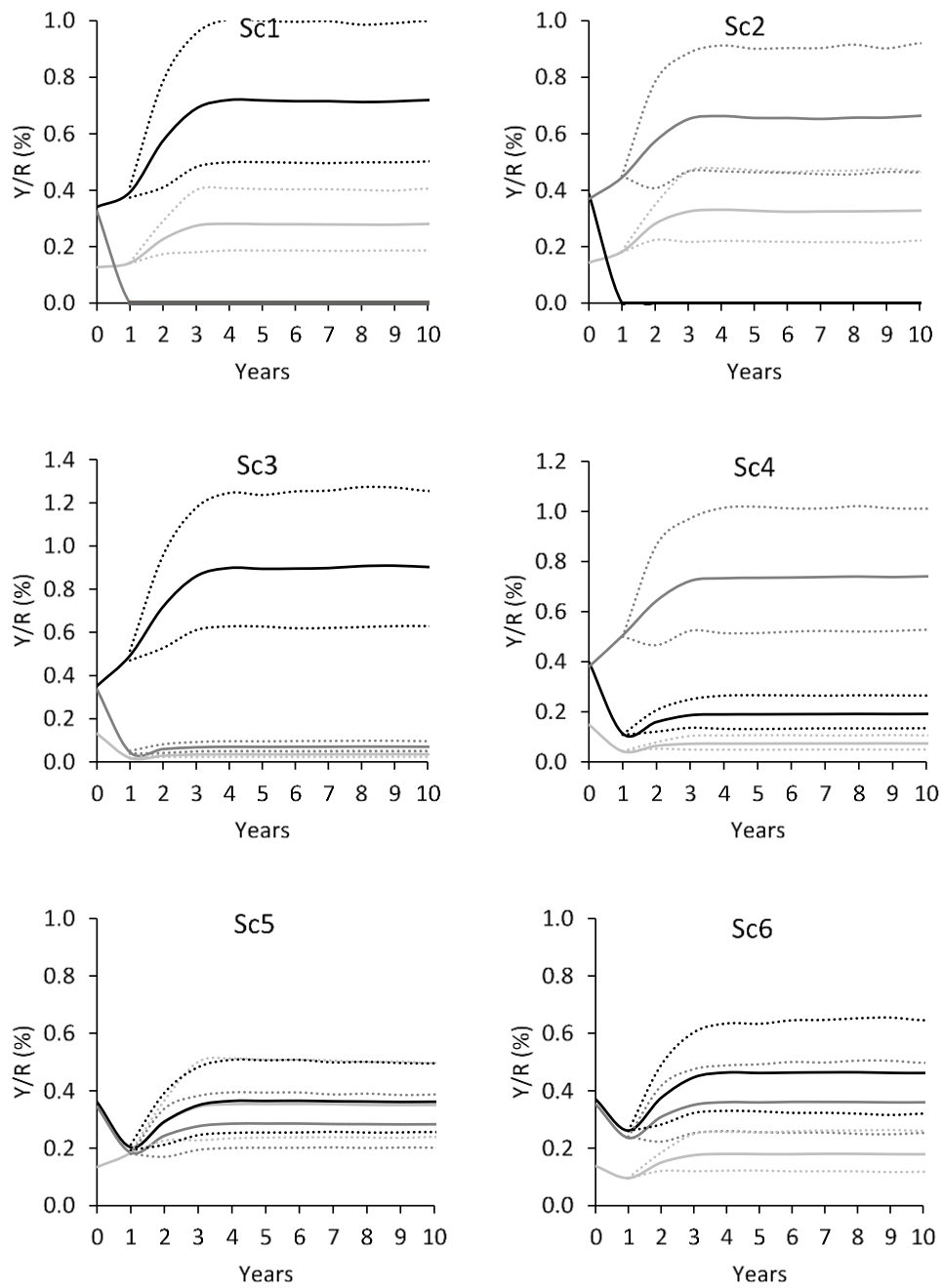


Fig. 21. Simulation of lane snapper yield per recruit (Y/R) by longline (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets, showing the mean and 95% confidence intervals of 1000 iterations (dotted line). Scenarios (Sc) are performed according to management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment models for the parameter set defined as  $L_{\infty}=64.94$ ,  $k=0.19$  and  $M=0.23$ . All scenarios accomplished reaching a relative SSB20% in a period of 8 years.

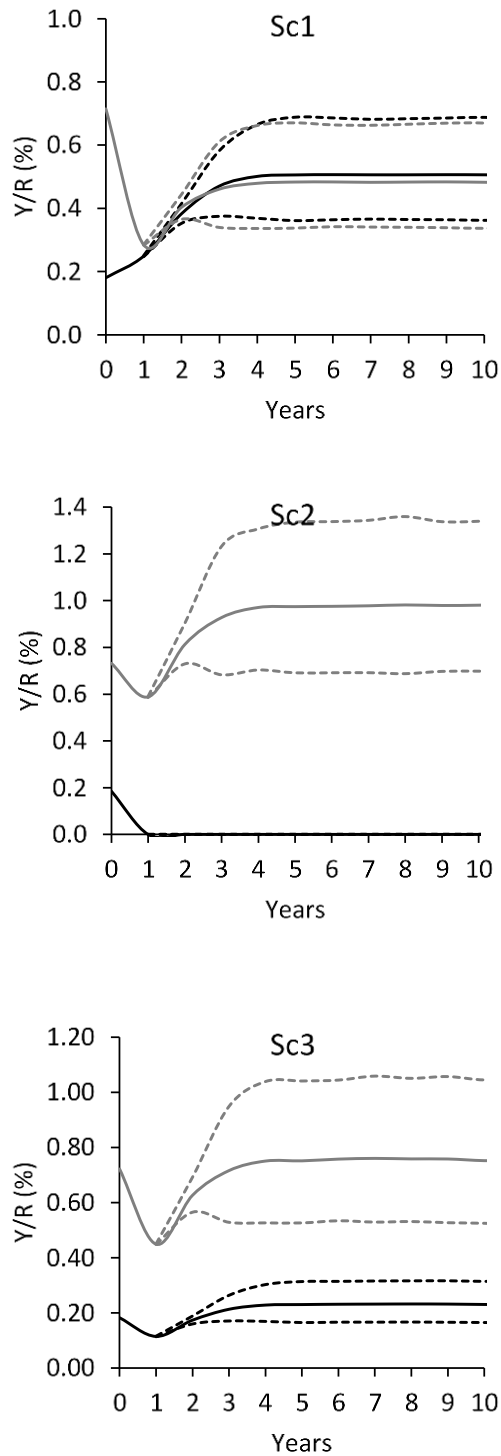


Fig. 22. Simulation of white grunt yield per recruit (Y/R) by longline (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets, showing the mean and 95% confidence intervals of 1000 iterations (dotted line). Scenarios (Sc) are performed according to management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment models for the parameter set defined as  $L_{\infty}=41.65$ ,  $k=0.27$  and  $M=0.15$ . All scenarios accomplished reaching a relative SSB20% in a period of 8 years.

### Determining population status

Fishery model parameterization under a data-poor framework is tied to several underlying assumptions (Leonart and Salat, 1997; Prince et al., 2015; Rätz et al., 2010). The *vBGP* obtained from fishery-dependent data in this study are probably biased because of continued overfishing that leads to age truncation (Kenchington, 2013; Saari et al., 2014). Furthermore, the fishery-dependent length frequencies used to estimate the growth parameters underestimated the smallest and largest fish. Additionally, indirect estimators spread the bias from the *vBGP* in calculating natural mortality (Kenchington, 2013; Then et al., 2015), which is important because population status depends on  $M$ . This is of great concern because inclusion of random data led to contradictory information about population status.

Few studies have compared the performance of indirect estimators obtained from “known” natural mortality rates of fish (Gislason et al., 2010; Kenchington, 2013; Newman, 2002; Then et al., 2015). Most argue that estimators based on  $T_{max}$ , mainly Hoenig’s estimator, should be selected instead of life-history correlation estimators (Kenchington, 2013; Then et al., 2015). Pauly’s and Ralston’s estimators, which are based on  $k$ , lead to higher  $M$  values (Kenchington, 2013). Additionally, Pauly’s estimator proved to perform badly for species that exhibit fast initial growth and later move toward slow growth, such as snappers (Kenchington, 2013). Then et al. (2015) found that the predicted error in the estimation of  $M$  using Pauly-based models was twice that using Hoenig-based models.

We found that Pauly’s estimator and similar estimators based on life-history correlations led to an underexploited or less exploited population status compared to Hoenig’s estimators, meaning that current effort could still be increased. This seems unfeasible because fishing mortality of small-sized fish between early and later years tended to increase, while landings and CPUE have been strongly reduced. The annual CPUE of the encircling gillnet vessels, including all caught species, was ten and eleven times higher than those of the bottom gillnet and long-line vessels, respectively. During the study period, the annual-average catch per vessel fell by 20.40% for encircling gillnets, by 48.06% for bottom gillnets and by 75.27% for the long-line vessels. These results support our findings of a growing trend toward overexploitation of both species.

When Gislason's *M-at-size* estimator was employed, variations in  $L_{\infty}$  and  $k$  led to large changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$  because both parameters play an important role in the estimation of natural mortality (Gislason et al., 2010). The lane snapper and white grunt, which are distributed around warm and temperate waters, exhibited quite different *M-at-size* values depending on the *vBGP*. The *M-at-size* estimator appears to perform poorly for species that exhibit large variations in  $L_{\infty}$  and  $k$ , so both uncertainty and bias associated with the calculation of *vBGP* have a strong impact on the estimation of  $M$ . Similarly, Abella's *M-at-age* estimator led to the greatest overexploitation status among the estimators tested. Abella's estimator appears to work well for temperate species with juveniles and adults spatially segregated (Abella et al., 1997), but underestimation of the  $M$  of tropical species with small and large individuals sharing the same areas could lead to overestimation of current fishery pressure. This would also lead to a large underestimation of fishing mortality in the long term (Clark, 1999).

The stock assessment is also conditioned by either the fast-growing or slow-growing hypothesis (Mellon et al., 2010). An increase in  $k$  means a reduction in longevity, affecting indirect estimates of natural mortality (ICES, 2014). Variation in the *vBGP* also led to large changes in the levels of  $F/F_{msy}$  and  $SSB/SSB_{max}$  for both species. Fast- and slow-growing have been assigned to the lane snapper and the white grunt distributed in the Caribbean Sea and East Atlantic Ocean (see Methods). However, great uncertainty remains in their estimates and comparisons because different methods and populations with diverse exploitation levels have supported the estimation of *vBGP*. Considering bias in the calculation of *vBGP* and  $M$  in our study, the stock assessments and management simulations were better resolved through testing both growth hypotheses. Our analyses showed that a higher  $k$  for lane snapper produced greater changes in the overexploitation level throughout the study period. Although not totally reliable, this  $k$  easily tracked changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$ . Greater changes between years would be desirable, given that the responses of the studied populations to unexploited or slightly exploited pressures are unknown.

Rindorf et al. (2016) stated that species of smaller asymptotic length (e.g., herring) do not acquire an inherently precautionary  $F_{msy}$  range, as occurs in species with a larger  $L_{\infty}$  (e.g., cod). Additionally, these authors mentioned that the  $SSB$  of stocks

with high asymptotic length are less prone to being affected by poor recruitment in some years. The reconstruction of virgin biomass and annual stock size obtained by our study was conditioned by 1) a steady state assumption (Rätz, et al., 2010), 2) an assumption of no changes in the selectivity by gear between years, and 3) a possible strong year-class effect in some years (as occurred in 2010 with one exceptional reconstructed white grunt catch). These conditions, along with the uncertainty in the estimation of the *vBGP*, probably contributed to masking the total effect of a landings decrease on the trend of stock size.

Punt et al. (2011) outlined among-stock penalties for assessing data-poor stocks when they are jointly caught with rich-data stocks, given that rich-data stocks are insensitive to penalties. However, the precision of outputs for data-poor stocks is greater when among-stock penalties are taken into account. Most of the stocks of small-scale fisheries in developing countries are data poor, meaning that the abovementioned approach cannot be implemented. From a derivation of the concept of “borrowing” data, as suggested by other authors (Kokkalis et al., 2015; Prince et al., 2015), we contrasted the best available local information and stock-related data from the literature to choose the least biased life-history parameters.

Overall, the evidence suggests that higher than natural fishing mortality occurred for highly targeted fish sizes. This finding is in agreement with the suggestion of using a conservative natural mortality in a data-poor context (Clark, 1999). Therefore, the estimators of natural mortality based on life-history correlations, which produced high values, were dismissed (Kenchington, 2013; Then et al., 2015). Although it cannot be claimed that the best data were found, plausible biological parameters of both species in the Central Guajira were introduced into the assessment models. The selected natural mortality rates (from 0.13 to 0.24 yr<sup>-1</sup>) produced overexploitation statuses for both species. Our findings indicate that the level of exploitation of stocks according to  $F/F_{msy}$  (e.g.,  $1 < \text{overfishing} < 3.5$  for lane snapper; Fig 5a) and  $SSB/SSB_{max}$  (e.g., overfished, 5-18% for white grunt; Fig 5b) were sensitive to changes in the *vBGP*. However, the *vBGP* tested in this study did not lead to contradicting population statuses for either species, as did the different estimators of natural mortality.

The problems of what particular level of overfishing and what spawning stock biomass remain under the current fishing pressure cannot be entirely addressed. Uncertainty analyses are increasingly used for some data-rich fisheries to avoid selecting the “best” parameters and to rather incorporate all plausible combinations of them (Scott et al., 2016). To implement this approach in data-poor fisheries, the quality and quantity of basic length information must be improved. Reliable model input data are required to separate bias from uncertainty. We suggest that tracking population changes in the trends of  $F/F_{msy}$  and  $SSB/SSB_{max}$  is a safer way of advising data-poor fisheries. This would avoid the management of data-poor fisheries based on the uncertain estimation of stock biomass and under a comprehensive range of probable fishing mortalities.

The stock assessment for lane snapper performed with the GOV data produced a higher overexploitation status than when the GOV and PAR combined data were used. A large underestimation of landings in rural areas for long-line and gillnet fleets by the government monitoring program yielded small variations in landings during the study period and provided a false view of the selectivity pattern for the lane snapper fishery. Managing this stock based on the government monitoring program increases the collapse risk of the lane snapper population, despite the fact that a higher overexploitation status has been indicated. This occurs because a stock assessment indicating a large overexploitation matched with low changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$  among years (Figure 5) would mean that the population timeframe observed (2009-2014) was quite stable, despite the fact that the stock is greatly overfished and the fishing pressure is unsustainable. This can lead to the false assumption that the lane snapper population can better absorb the threats of overfishing. Conversely, by including data from participatory monitoring, we highlight that in nine years, the  $SSB/SSB_{max}$  could have been reduced by half while  $F/F_{msy}$  could have been doubled. Therefore, in data-poor fisheries, the overestimation of fishing mortality (Thorson and Prager, 2011) does not necessarily decrease the risk in management advice.

The stock assessment of the white grunt did not lead to distinct outputs of  $F/F_{msy}$  and  $SSB/SSB_{max}$ , regardless of which data source was used. This is because the

encircling gillnet fleet dominated the species harvesting and the selectivity pattern for this fleet matched that of the gillnet fleet. Therefore, distinct landing points recorded by either monitoring program did not alter the selectivity pattern for the white grunt fishery. Nevertheless, the biomasses removed among fleets were quite different, implying that reaching a maximum sustainable yield ( $F/F_{msy}=1$ ) would require that the biomass harvesting power by fleet be taken into account. This, however, is only feasible if landings recorded by the participatory monitoring program are considered in the stock assessment and further management simulations.

We found that the uncertainty associated with the different biological parameters and stock recruitment models used to perform each management scenario was moderate (lane snapper, 9%-33%; white grunt, 29%-40%). Consequently, the uncertainty identified during the definition of population status was considerably reduced when all the management scenarios were tested. According to our findings, the populations of lane snapper and white grunt are largely overexploited, meaning that recruitment may be compromised. The constant recruitment model included the underlying assumption of little or no effect of depleted stock size on recruitment (Brooks et al., 2010). However, it is also possible that the number of recruits per spawner increase at low population abundances (Jennings, 2001), with the possibility of a quasi-linear curve. Thus, irrespective of the actual stock-recruitment model and which life-history parameters correctly describe the two species, the management scenarios from a data-poor stock assessment could offer useful advice on a reasonable fishing effort reduction range (e.g., vessels per fleet). However, findings from stock assessment and management scenarios relate to the specific conditions of evaluation (Carruthers et al., 2014), so for providing management advice, the parameters of the model should remain constant over time.

### **Conclusions**

The government monitoring program led to a misleading description of landings and fishing mortality. This triggered riskier definitions of population status and of the overall fishery trend. The incorporation of data from the participatory monitoring program offered a better understanding of the exploitation patterns for the lane snapper and white grunt populations. Our findings suggest that sensitivity tests should

become part of the advice for data-poor fishery management, as understanding the bias in the data for model parametrization cannot be overlooked. In addition, several limitations underlie the single-species stock assessment in a data-poor context. Some of them can be addressed (e.g., the definition of population status), but others cannot (e.g., absolute values of SSB). Therefore, control of fishing effort should be preferred for management instead of total allowable catch. Finally, our results support the idea that the decentralization of fishery management structures and the participation of local communities are required to obtain improved data. If stock assessments produce a more reasonable population status for key species, reliable advice can be offered for participatory management of small-scale fisheries in data-poor contexts.

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2.4. Exploring uncertainty in life-history parameters  
when applying data-limited methods for stock  
assessment



**Exploring uncertainty in life-history parameters when applying data-limited methods for stock assessment**

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**Abstract**

Accuracy in length-based stock assessment outputs of data-limited species is usually hindered by the lack of knowledge of biological parameters. We address how to incorporate life-history parameter uncertainty into the estimation of spawning potential ratio (SPR). The Peruvian grunt (*Anisotremus scapularis*) fishery is as a case study for low resolution in biological and stock status information. Length-based and hierarchical methods, in addition to literature information, were used to estimate nine von Bertalanffy asymptotic lengths ( $L_{\infty}$ ) and growth coefficients ( $k$ ). Maximum age, growth parameters and other life history information was used to empirically derive 14 indirect estimates of natural mortality. This resulted in 118  $L_{\infty}$ - $M/k$  combinations used as scenario inputs into the length-based method to estimate SPR from the length structure of the stock (LB-SPR). Three treatments of uncertainty in the parameter inputs or outputs used: 1) equal contribution of the 118 individual SPR outputs, 2) triangular distribution of the LB-SPR model inputs and 3) equal probability contribution of LB-SPR

model inputs. Our results suggest that the advice derived from SPR estimates needs to incorporate uncertainty in parameter inputs as well as the internal estimate of uncertainty in SPR, and should avoid considering only the central tendency of the output.

### **Keywords**

Life-history parameters, uncertainty, data-limited fisheries, Peruvian grunt, LB-SPR

### **Introduction**

Small-scale fishers are often identified as key players in the recovery of overexploited fish and invertebrate stocks supplying food for rural people and contributing to healthy marine ecosystems (Pauly, 2018; Pitcher and Cheung, 2013; Teh and Pauly, 2018; Teh et al., 2017). Stocks harvested mainly in small-scale fisheries tend largely to be unassessed and exhibit poorer status than unassessed large stocks (Costello et al., 2012). Data-limited stock assessment methods able to determine stock status and/or derive catch or effort control recommendations are widely applicable, greatly needed, and increasingly being developed and tested (Carruthers et al., 2014; Chrysafi and Kuparinen, 2015). A subset of the greater data-limited toolbox are methods that provide stock status for fished stocks that do not have historical time series of catches, but do have length compositions (Hordyk et al., 2015b; Kokkalis et al., 2015; Rudd and Thorson, 2017). These methods are particularly important, as removals are often either historically unavailable or difficult to monitor.

Length-based SPR (Hordyk et al. 2015b) is a notable length-based assessment approach gaining attention (Rudd and Thorson 2017) and application (Prince et al. 2015b) that uses length composition data and life-history parameters (Leonart and Salat

1997; Prince et al. 2015b) to estimate transient spawning potential ratio (Beckensteiner et al., 2016). Two of the major inputs to that model, the von Bertalanffy Growth Parameters (*vBGP*)  $L_{\infty}$  (asymptotic length) and  $k$  (the growth coefficient), and natural mortality ( $M$ ) are frequently unavailable or poorly estimated (Prince et al., 2015b). For this reason, the LB-SPR approach is parameterized to use the  $M/k$  ratio to benefit from the taxonomically-conservative characteristics of that ratio (Prince et al. 2015a). Despite the general qualities of  $M/k$ , a range of values is not always obvious, thus calculating and incorporating uncertainty in species-specific  $M$  and  $k$  values can be a powerful way to address parameter bias and imprecision.

Data-limited fisheries are typically reliant on fishery-dependent information and often only length frequency data, for estimating *vBGP* parameters (Gayanilo et al., 2005; Mildenerger et al., 2017; Taylor and Mildenerger, 2017). Highly exploited populations, however may exhibit a truncated length structure (Saari et al., 2014), leading to an underestimate of  $L_{\infty}$  and overestimate of  $k$ . Length-based estimates of *vBGP* are also affected by gear selectivity (e.g. logistic vs dome shaped; ( Gwinn et al., 2010). Thus, gear selectivity may not be a representative sampler of the underlying population length variability, leading to uncertainty in estimation of growth parameters.

Natural mortality is typically even more difficult to directly measure for most stocks (Scott et al., 2016). In lieu of direct measurement, life history parameters such as maximum age and *vBGP* parameters are commonly used to feed empirical estimators of natural mortality (Then et al. 2015), thus retaining correlational  $M/k$  relationships, but making them species-specific. Kenchington (2014) found that there is no one  $M$  estimator that works well for all taxa. Additionally,  $M$  estimators based on maximum estimated age show somewhat better performance than those based on other life-

history correlates (Kenchington, 2014; Then et al., 2015), but still involve large uncertainty (e.g., What maximum age to choose? How big was the ageing error determining the maximum age?).

Many data-limited species also lack species-specific information for getting life-history parameters or available data for performing length-based analysis. For instance, although the Peruvian grunt (*Anisotremus scapularis*) plays a key role in the local economy and food supply of rural communities,  $L_{\infty}$ ,  $k$  and  $M$  are not available in the global database FishBase (Froese and Pauly, 2015). Kokkalis et al. (2015) and Prince et al. (2015b) suggested that life-history information from related-species could be used for parameterizing data-poor stock assessment models (the “Robin Hood” approach of Punt et al. (2011)). Recently, Thorson et al. (2017) developed a hierarchical approach that considers taxonomic structure (FishLife) for estimating the life-history parameters of fishes, allowing the borrowing of  $M$  values from higher taxonomic levels when not available at the species level.

Determining a reasonable stock status of data-limited species may ultimately be hindered by the values and uncertainty in life-history parameters applied (Hordyk et al., 2015a; Kokkalis et al., 2017). Particularly, uncertainty related to  $vBGP$  and  $M$  show large sensitivities in many approaches and should be carefully considered (Gwinn et al., 2010; Ramírez et al., 2017; Then et al., 2015). Here we look to advance the use of confronting uncertainty in life history parameters by quantifying the impact of such uncertainty when applying the LB-SPR method to the Peruvian grunt (*Anisotremus scapularis*).

We estimated nine sets of  $L_{\infty}$  and  $k$  values using a variety of length-based estimators of the  $vBGP$ , and also explored borrowing values from closely related taxa using estimates derived from both FishLife and a congener (the Burrito grunt,

*Anisotremus interruptus*). We then estimated  $M$  using 14 different empirical estimators (Table 11). This resulted in 118  $L_{\infty}$ - $M/k$  combinations to assume plausible life-history parameter values of the Peruvian grunt as input into the LB-SPR method to estimate stock status. We define and apply the life-history parameters distributions in a variety of forms to estimate SPR, then ultimately discuss our findings in regard to incorporating uncertainty in the estimation of stock statuses for data-limited resources.

## Methods

### Case study: Peruvian grunt fishery

The Peruvian grunt inhabits mostly beach-break zones (Angel and Ojeda, 2001; Perez-Matus et al., 2007), but official monitoring is concentrated in fishing ports, leading to the beach-bank fishery to be undersampled. In order to collect samples representing the length composition, fishery-dependent information was gathered through the monitoring biodiversity program performed by PERU LNG and the Smithsonian Conservation Biology Institute from 2012-2015 (Alonso et al., 2013).

Six rocky and sandy beach locations in the coastal border of the Departments of Lima and Ica were sampled over seven-day campaigns in autumn and spring. Sampling was carried out in the same place and time that fishers conducted their activities (including at night) in order to capture the dynamic of beach-fishing operations. Size and weight were recorded for 2754 individuals caught in all hauls by the beach fishery gears. Length distribution in autumn and spring of every year were merged to increase the length sample size as recommended by Prince (2017). Given that the other two seasons were not sampled and there is not information on fishing effort over those periods, we assumed that the sampled length structure represented properly the exploited stock over the whole year, and that the selectivity over the two sampled season is the same.

### Estimation of life-history parameters

698 individuals were dissected to assign macroscopically a gonadal stage based on a scale with four maturity phases: immature (phase I), developing or regenerating (phase II), ripe (phase III) and spent (phase IV) (Palazón-fernández et al., 2007).

Subsequently, combined –sex and sex-specific size at maturity curves were fit to a logistic curve (Figure S1). Parameter estimates include size at which 50% ( $L_{50}$ ) and 95% ( $L_{95}$ ) of all sampled fish were sexually mature. The LB-SPR model uses both female maturity parameters.

No ages were available, so four length-based methods were used to estimate  $vBGP$ s: 1) Shepherd's method using Bhattacharya fit to a non-linear model (Pinheiro et al., 2015); 2) *slca* method (fishmethods r-package, Nelson and R-Core Team 2017) 3) ELEFAN GA method and 4) ELEFAN SA method (TropFishR r-package, Mildenerger et al. 2017; Taylor and Mildenerger 2017). Each set of estimates was assumed equally possible to represent growth of the Peruvian grunt in nature, but of high uncertainty given the lack of age data.

We also used methods that did not require raw data estimation. FishLife (Thorson et al., 2017) is a hierarchical random-effects model that considers taxonomic structure in the estimation of life-history parameters using inputted life history values from FishBase (Froese and Pauly, 2015). FishLife was used to obtain  $L_{\infty}$  and  $k$  for the Peruvian grunt, but given the  $vBGP$  of this species are not included in FishBase, those parameters were not available at the species level. When this is the case, the FishLife tool estimates parameters from the nearest taxonomic neighbor, in this case the Burrito grunt (*A. interruptus*), the only other species within the genus with age and growth information (predictive routine). FishLife also allows updating estimates with new information, so we used these updated estimates of life-history parameters by combining the global data and local estimates based on the length methods (update routine). Additionally, we use estimates of  $L_{\infty}$  and  $k$  of Burrito grunt provided by FishLife. Lastly, we borrowed values for the Burrito grunt from each of the two available literature sources (Cruz-Romero et al., 1993; Espino-Barr et al., 2004) and used them as separate estimates of life history values. This represents the poorest way to obtain the life-history parameters for the Peruvian grunt and assumes that the biology and ecology of both species are quite similar and their responses to fishing pressure are comparable.

The value of  $M$  is typically a highly sensitive input to most stock assessment methods (Ramírez et al., 2017; Scott et al., 2016), but the “true” value of  $M$  is usually unknown (e.g., can be biased) and uncertain (can be imprecise; Hunter et al., 2015;

Kenchington, 2014). Given no direct measures of  $M$  are available, and instead of using one estimate of  $M$ , 14 empirical estimators of  $M$  were used to estimate values and uncertainty in  $M$  (Table 11) using the web-based Natural Mortality Tool (Cope 2017). In addition to using values such as longevity and age at maturity, all nine sets of  $vBGP$  were also used to estimate  $M$ . Additional inputs are listed in Table 11. Estimates of  $M$  were also derived from FishLife for burrito grunt (closer species) and Peruvian grunt using the predictive and update routines.

All related  $M$  and  $k$  value were subsequently formulated as  $M/k$ , resulting in 118  $L_{\infty}$ - $M/k$  combinations so as to maintain parameter correlation structure (Table 11).

Table 11. Input data used to parameter the life-history parameters uncertainty of the Peruvian grunt

Package/method	FISAT II (Shephard)	Fishmethods (Shephard)	TropFishR (ELEFAN GA)	TropFishR (ELEFAN SA)	FishLife (closer sp)	FishLife (Predictive)	FishLife (Updated)	Burrito grunt (Espino-Barr et al., 2004)	Burrito grunt (Cruz- Romero et al., 1993)
$L_{\infty}$ (cm)	66.37	65	60.2	58.11	57.62	50.45	62.05	50.59	48.2
$k$ ( $y^{-1}$ )	0.23	0.18	0.18	0.16	0.2	0.23	0.18	0.15	0.13
to (y)	-0.47	-0.55	-1.94	-0.92	-	-	-	-0.92	-0.01
Max. estimated age (y)	7.5	10	12.5	18.5	15.53	13.69	17.88	22	22
$L_{50}$ (cm)	21.2	21.2	21.2	21.2	28.1	25.27	31.59	31	31
$L_{95}$ (cm)	35.87	35.87	35.87	35.87	35.87	35.87	35.87	38	38
$L_{50}$ (y)	1.25	1.75	2	0.5	3.69	3.23	4.19	5.75	8
$L_{95}$ (y)	3	3.75	3.25	5.25	4.25	4.75	4.25	8.75	12
Asym. Weight (g)	3565	3565	3565	3565	2705.39	1723.31	2827	5051	5051
Water temp. (°C)	19	19	19	19	26.01	25.35	19.66	24	24
$M$ (based on)									
Then_1 (Age max.)	0.77	0.59	0.48	0.34	0.4	0.44	0.35	0.29	0.29
Then_2 (Age max.)	0.68	0.51	0.41	0.28	0.33	0.37	0.29	0.23	0.23
Then_3 (Age max.)	0.73	0.54	0.43	0.29	0.35	0.4	0.3	0.25	0.25
Hamel (Age max.)	0.72	0.54	0.43	0.29	0.35	0.39	0.3	0.25	0.25
AnC ( $vBGP$ )	0.76	0.55	0.4	0.24	0.27	0.3	0.23	0.18	0.2
Then ( $vBGP$ )	0.16	0.14	0.14	0.13	0.16	0.18	0.14	0.13	0.12
Jensen_1 ( $vBGP$ )	0.33	0.27	0.26	0.23	0.3	0.35	0.27	0.22	0.2
Jensen_2 ( $vBGP$ )	0.35	0.29	0.28	0.25	0.32	0.37	0.29	0.24	0.21
Pauly ( $vBGP$ +temp)	0.44	0.39	0.39	0.37	0.5	0.56	0.4	0.41	0.38
Gislason ( $vBGP$ +temp)	0.25	0.2	0.18	0.15	0.19	0.18	0.19	0.19	0.15
Chen-Wat ( $vBGP$ + max.)	0.42	0.34	0.26	0.24				0.22	0.25
Roff (Age maturity)					0.55	0.63	0.48	0.33	0.21
Jensen_Amat (Age maturity)					0.45	0.51	0.39	0.29	0.21
Lorenzen (weight)	0.27	0.27	0.27	0.27	0.27	0.27	0.27	0.26	0.26
FishLife package					0.46	0.54	0.38		

## Estimation of the Spawning Potential Ratio

We focused on applying the LB-SPR method to the Peruvian grunt stock data to explore the impact of the uncertainties of life-history parameters  $L_\infty$  and  $M/k$  on the SPR estimates, as well as offer an example of what to do when only length data and limited life history parameters are available (a common condition of many data-limited situations). LB-SPR assumes, among others, an equilibrium reference (stable spawning potential SPR under no fishing), constant recruitment, asymptotic selectivity and a constant fishing mortality (Prince et al., 2015b). Potential intra-annual changes in length-frequencies distribution are therefore not influenced by variation in recruitment, but rather fishing mortality or selectivity. To interpret the meaning of the SPR estimate,  $SPR_{0.4}$  was used as a precautionary target referent point given the lack of knowledge of the stock recruitment relationships (Myers et al., 1994; Thorson et al. 2012), while  $SPR_{0.2}$  was used as a limit reference point (Rosenberg, 1993). Multiple ways were explored to determine which values of  $L_\infty$  and  $M/k$  would be used to calculate SPR.

Three treatments of  $L_\infty$ - $M/k$  uncertainty were used to quantify SPR uncertainty. The first approach applies the 118  $L_\infty$ - $M/k$  combinations in turn to the LB-SPR model, resulting in 118 SPR outputs that were then resampled 10,000 times to produce a distribution of SPR values.

A second approach assumed a triangular distribution for each parameter pairing, with the lowest, highest and median pairing values of the 118  $L_\infty$  and  $M/k$  pairings used to set this distribution from 10,000 draws. The triangular distribution is commonly applied in data-limited stock assessment methods (Hordyk et al., 2015a; Scott et al., 2016). Another triangular distribution constrained to the minimum and maximum values of  $L_\infty$  and  $M/k$  at the first and third quartiles was also considered. This removed the influence of possible outliers in defining the distribution.

Lastly, a 10,000 draws of the 118  $L_\infty$ - $M/k$  pairings were taken, and run through the LB-SPR approach. The overall variance was then calculated by first retaining the within run estimate of variance (which is sensitive to length frequency sample size) for each of the 10,000 SPR estimates. All 10,000 estimates for variance were then combine

to get an overall estimate of SPR variance. For all parameter treatments, the remaining life history input value, the  $L_{50}/L_{\infty}$  ratio, was assumed constant at 0.3648.

## Results

### Life history values

The number of sampled individuals in 2012 (n=498), 2013 (n=663), 2014 (n=1021) and 2015 (n=572) provided sufficient sample size to fit the LB-SPR model (Fig. 23). The median size of the Peruvian grunt caught by the beach artisanal fishers ranged from 26.5 cm to 28.5 cm. Size at first maturity for male and females were comparable ( $L_{50} \sim 21$  cm) (Figure S1), but  $L_{95}$  was reached later by males (41.21 cm) than by females (35.87 cm). Overall, this fishery targets mature individuals with no decrease in selectivity, so likely fulfills the LB-SPR assumption of asymptotic selectivity

Given that SPR is defined by the age truncation of the stock (here observable from maximum caught sizes relative to asymptotic length), defining  $L_{\infty}$  can be a highly influential parameter in the estimate of LB-SPR. Length-based estimates of  $L_{\infty}$  were larger than 58 cm, while those estimates based on FishLife ranged from 50.45 cm (based on borrowed Burrito grunt values) to 62.05 cm (updated with the Peruvian grunt length-based estimates) (Table 11). The FishLife results suggested that the absence of specific information on life-history parameters for Peruvian grunt in FishBase may inject bias when using those parameters in a model.

Natural mortality largely varied depending on which estimator was considered (Table 11). Values derived from the  $L_{\infty}$  and  $k$  parameters ranged from 0.223 to 0.563.  $M/k$  ratios ranged from (1.63-2.02), substantially different from assumption that  $M/k$  is approximately 1.5 (Froese et al., 2018a, 2018b).

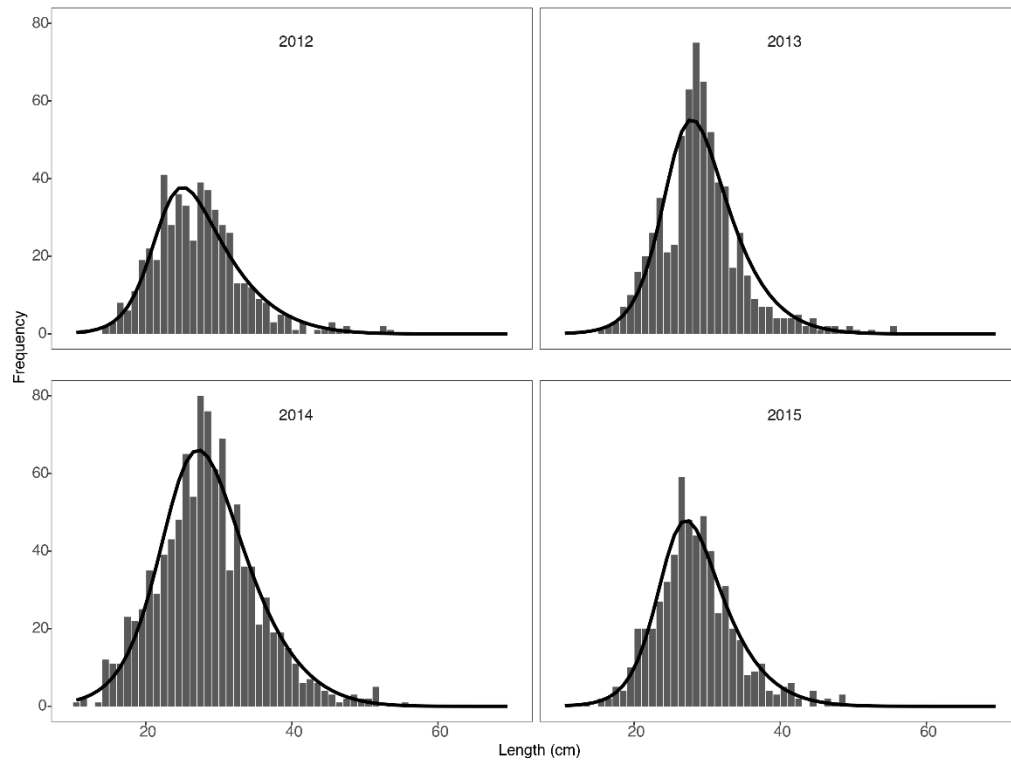


Fig. 23. Length-frequency structure of the Peruvian grunt stock fitted (black line) by the LB-SPR model.

#### SPR estimates

The SPR distribution obtained under the assumption of proportional contribution of the 118  $L_{\infty}$ - $M/k$  combinations produced two main peaks around 0.10 and 0.25 for the four years (Fig. 24a). Both modes indicate the Peruvian grunt stock is below the target (SPR < 0.4, Myers et al. 1994), but is not clear whether the stock was below or above the limit reference point of  $SPR_{0.2}$ . The attribute of multiple modes was common in these results (Fig. 25, S2, S3 and S4), and values higher than 0.4 were linked to the Burrito grunt life-history parameters obtain from FishLife.

When life-history parameters were adjusted to triangular distribution (Fig. 26), the SPR distribution acquired a unimodal shape (Fig. 24b and c). Regardless which criteria was used to define the minimum and maximum distribution values, the SPR distribution tended to be below 0.2. The overall distribution of SPR values was expectedly more precise when imposing the more limited minimum and maximum  $L_{\infty}$  and  $M/k$  values.

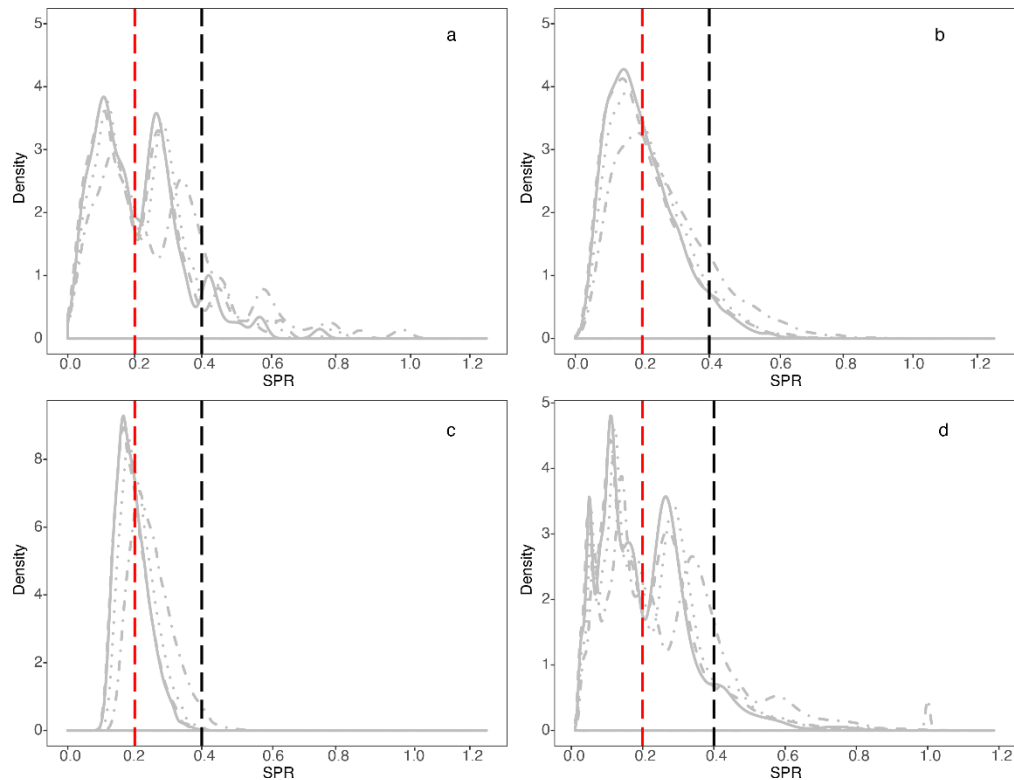


Fig. 24. SPR distribution derived from a) equal probability contribution of all life-history parameter sets, b) triangular distribution of life-history parameters, c) triangular distribution after constraining the minimum and maximum values of  $L_{\infty}$  and  $M/k$  to the first and third quartiles and d) Monte Carlo simulation. The LB-SPR analysis was performed in 2012 (dashed line), 2013 (dotted line), 2014 (dotdash line) and 2015 (solid line). Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

Using 10,000 draws of the  $L_{\infty}$ - $M/k$  pairings also led to  $SPR < 0.4$  (Fig. 24d). However, this analysis revealed three SPR peaks below  $SPR_{0.2}$ . The SPR distribution also indicated very low probability that the Peruvian grunt stock SPR is  $> SPR_{0.4}$ . Low probability above the target SPR value again derived from the Burrito grunt life-history parameters.

The updated routine of FishLife, which estimated the life-history parameters from both global (FishBase) and local (length-based) data, produced the most consistent SPR distributions (Fig. 27, "Peruvian grunt-updated" panel).

The SPR distribution was also influenced by the contribution of large individuals in the perceived stock structure. The SPR distribution in 2014 (Fig. 27), which reported larger individuals than other years, was more scattered than in 2012 (Fig. S5), 2013 (Fig. S6) and 2015 (Fig. S7).

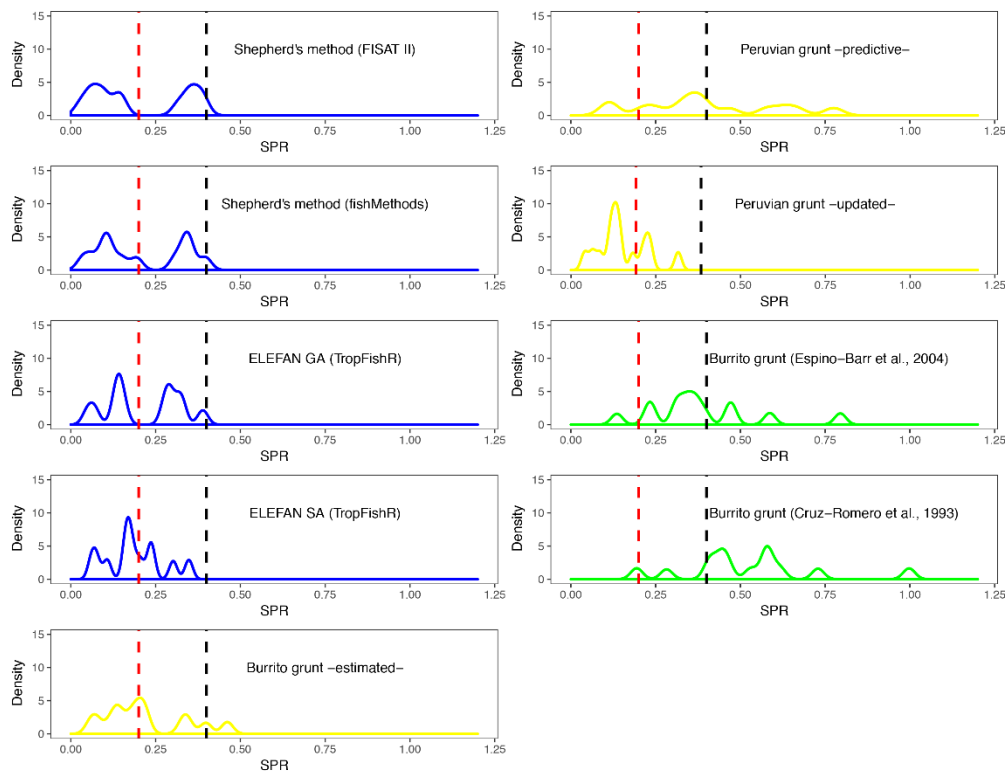


Fig. 25. SPR distribution in 2014 produced by proportional contribution of the life-history parameters derived from the nine different methods to estimate the  $vBGP$ . Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

## Discussion

The impact of the uncertainty in life-history parameters on the stock status of a data-limited species was shown to be large and important to interpretation stock status. Our study revealed that the SPR estimates can be far more uncertain than the within model estimates would indicate, sensitive to the life-history parameters used and underlying assumptions that define parameter distribution.

It is unlikely to accurately identify which of the 118  $L_{\infty}$ - $M/k$  combinations used in this study is the “best” candidate to describe the Peruvian grunt stock. The length-based estimates of  $vBGP$  on a data-limited context propagate the bias that may underlie the fishery-dependent length data. Ongoing overfishing may lead to age-truncated populations, implying, for instance, underestimation of asymptotic length (Kenchington, 2014; Ramírez et al., 2017; Saari et al., 2014). In these circumstances the perceived  $k$  is overestimated, thus possibly impacting estimates of natural mortality as well. The Peruvian grunt fishery mainly harvested adults using several kind of gillnet and hook

sizes. Although fishers recognized that medium and small sizes are recently more targeted than some years ago (John Ramírez, Comm. Pers.), there is no historical information to determine changes in gears selectivity. Consequently, the *vBGP* here estimated from the different length-based models may provide an incomplete picture of the growth curve uncertainty.

Length-only estimators of *VBGP* are subject to bias and imprecision becoming them suboptimal choices when fitting growth curves. Even if the assessed stock holds few biological and fishery information (i.e. data-poor species) and inhabit an environment that hampers discriminating the growth between seasons (Newman et al, 2000), ageing methods based on otolith readings may contribute to reduce the uncertainty related to length-based estimates of *vBGP*. Further uncertainty in *VBGP* could be explored by hypothesizing alternative selectivity patterns, though changes in selectivity in LB-SPR are currently limited to changes in logistic selectivity a composite selectivity curve that weighs different periods of selectivity and combines data across years could be explored.

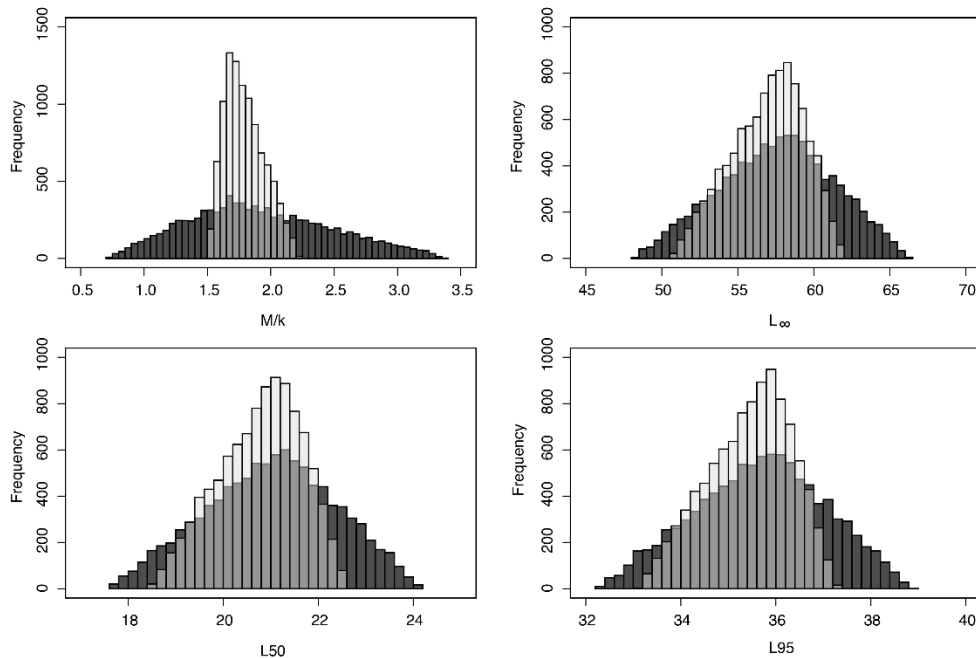


Fig. 26. Uncertainty in life-history parameters incorporated into *SPR* estimates through triangular distribution. Distribution values defined by the maximum, minimum and median values estimated for each parameter (dark grey bars). Distribution values of the minimum and maximum values of  $L_{\infty}$  and  $M/k$  constrained to the first and third quartiles (light grey bars).

Despite natural mortality's importance in influencing the sensitivity of most population models, direct measures of this parameter are rare (ICES, 2014; Scott et al., 2016). Thus,  $M$  is often estimated by indirect methods from life-history parameters that are comparatively easier to estimate, such as longevity and  $vBGP$  (Gislason et al., 2010; Kenchington, 2014; Then et al., 2015). Biases incorporated through the  $vBGP$  estimation can be propagated to  $M$  estimates, leading ultimately to bias in the stock assessment outputs and exploitation rates for maximum sustainable yield (Gwinn et al., 2010; Ramírez et al., 2017). Our results indicated that the definition of the stock status of the Peruvian grunt depends on the treatment of the range in possible life-history parameters. Natural mortalities estimated by 14 indirect estimators produced a wide distribution of SPR estimates (Fig. 25 and Fig. 27). Our example here provides an outline on how to properly treat uncertainty in those correlated life history parameters rather than just picking one set of values and relying on within method uncertainty to sufficiently describe derived output uncertainty. This is particularly true with LB-SPR, as the within model uncertainty is based on length counts, thus are very small with highly sampled length distributions.

An equal contribution of all  $L_{\infty}$ - $M/k$  combinations in the LB-SPR model did not require defining the “right” life-history parameters distribution. Conversely, it was accepted that all methods used to estimate  $vBGP$  and all indirect estimators of  $M$  may produce reasonable information of the Peruvian grunt in nature. For the Peruvian grunt, this wide exploration of uncertainty mainly demonstrated a greatly reduced reproductive potential. The lowest SPR peak (0.1) is plausibly produced by a population with an  $M/k$  value closer to the expectation from Beverton and Holt life-history invariant theory ( $M/k=1.5$ ; Hordyk et al., 2015a). The second SPR peak (0.25) is produced by a population with higher productivity ( $M/k=2.2$ ). The estimated life-history parameters for the Peruvian grunt tend to demonstrate higher productivity ( $M/k>1.5$ ). However, there is no evidence to argue one SPR peak over the other. As the stock looks greatly reduced despite relatively high productivity, the lower peak (e.g.,  $SPR_{2014} < SPR_{0.2}$ ) may be considered as a precautionary status, while higher peak (e.g.,  $SPR_{0.2} < SPR_{2014} < SPR_{0.4}$ ) will likely lead to less-drastic management measures for recovering the SPR.

Each state of nature should be considered thoughtfully when applying management measures to this species.

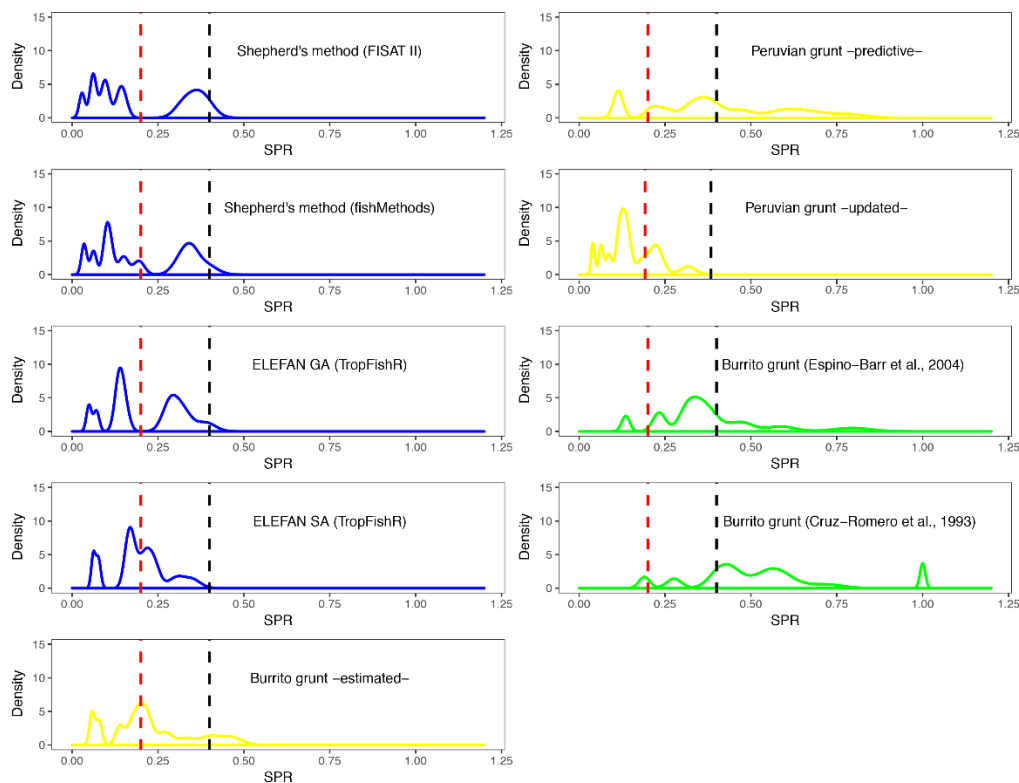


Fig. 27. SPR distribution in 2014 obtained from the life-history parameters distribution uncertainty estimated by the Monte Carlo simulation. Length-based (blue lines) and FishLife (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

The LB-SPR model assumes asymptotic selectivity (Prince et al. 2015b). This means that non-recorded adult sizes are interpreted as already removed by the fishery. The latter is a strong assumption that is often not realized in small-scale fisheries. Most of fishing gears used by these kind of fisheries (e.g., hooks and gillnets) can exhibit dome-shape selectivity (Huse et al., 1999; Kuparinen et al., 2009). The Peruvian grunt fishery mainly targeted adult classes, thus, potential biases in the SPR outputs by either a misleading representation of the stock length structure or dome-shaped selectivity were considered low. If however, all adult-size classes are not properly sampled, SPR output could be underestimated.

Neither the simple calculation of SPR from all  $L_{\infty}$ - $M/k$  combinations nor the triangular distribution may properly account for uncertainty in estimates of life-history parameters. In the first case, extreme life history pairing (e.g., small  $L_{\infty}$  and high  $k$ ) are

weighted equally to more distribution-centered values. The second case requires assuming a certain distribution well defined by its peak, minimum and maximum values. None of these two assumptions may be appropriate to explain the life-history parameters distribution of the Peruvian grunt. The final treatment of resampling with replacement all life-history pairings used to set the LB-SPR model offered a SPR distribution that allowed the 118 models to define the natural distribution of the uncertainty. The contribution of input data that led to unlikely high SPR (reflecting a stock in good shape and able to support higher fishing rates) was diminished. Thus, the resultant parameters pairing distribution emphasized the more diminished SPR condition of the Peruvian grunt.

Our findings suggest that borrow life-history parameters of congeners (e.g. *A. interruptus*, the Burrito grunt) may lead to greatly different results. Given that the Burrito grunt exhibit a smaller  $L_{\infty}$  than the Peruvian grunt, the LB-SPR model indicated a significantly more optimistic SPR above the target reference point ( $>SPR_{0.4}$ ), while all species-specific estimates were well below the target and even the limit. Additionally, some estimates of  $M$  were dependent on the  $vBGP$ , and therefore also influenced by borrowing information from a congener. Eight of the nine  $vBGP$  sets led to stocks below the target ( $SPR < 0.4$ , Fig. 27), with the only one above being that borrowed from the Burrito grunt. Given that empirical estimates of  $M$  depend on what  $vBGP$  are used to estimate natural mortality, to borrow information of congeners to set the LB-SPR model may increase the uncertainty in the SPR, and in this case, in a biased way. The updated estimates from the FishLife package did produce a more unimodal SPR distribution ( $0.05 < SPR < 0.25$ ), suggesting caution when borrowing even from closely related species. We therefore submit the Peruvian grunt as an example of where borrowed life history parameters can produce notably different results. High value should be placed on species-specific data collection, however minimal it may be, to at least compare maximum sizes to ground truth borrowed species  $L_{\infty}$ .

Despite length data limited to 2012-2015, fishery-dependent data reported larger individuals in 2014 compared to other years. In terms of the stock assessment, a higher contribution of larger individuals to the stock structure led to larger SPR dispersion. This result means that the presence of older individuals emphasize the

impact of the underlying assumption regarding the stock structure of the unfished stock (i.e.  $M/k$  higher or lower than 1.5). One treatment of limited multi-year data could be to combine them over all years in order to avoid vagaries in any given year, though this should be done in a way that first looks for patterns in the data (e.g., quickly dropping, then rising SPR values that cannot be explained by catch history) to exclude the possible complications of recruitment events.

### **Conclusions**

Life history parameters are important sources of uncertainty in most stock assessment models. The exploration of estimating multiple values and combining the internal model estimates with the among life history parameter model estimates produce a greater estimation of uncertainty in a data-limited length-based method (LB-SPR), and clearly demonstrated the benefit of calculating and reporting model specification uncertainty, not just the within model estimate of uncertainty, which does not consider model misspecification. Implicit in this activity is understanding where the values being used to define parameter uncertainty comes from (e.g., directly estimated, previously estimated species-specific value, or borrowed for a close taxonomic relative) and how that uncertainty is incorporated into a prior distribution so it can be perpetuated into the estimation process. This fuller treatment of uncertainty often also needs accompanying definitions on how to apply risk to the interpretation of such uncertainty (Smith et al. 1993 and papers therein). The investment in species-specific life history parameters can often reduce the uncertainty to give a more accurate picture of stock status, though a large amount of imprecision may still remain when life history parameters are empirically, rather than directly, estimated. Specifically, these findings support greater investment in obtaining ages for Peruvian grunt in order to improve the age and growth estimation. Such improves would feedback into improved indirect methods for estimating natural mortality, and consequently, promoting a better definition of the stock status without sacrificing the characterization of uncertainty.

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## 2.5. Summary of result



## Results chapter 2.1

### Estimating fishing mortality and stock size

The landings of hake in the Northwestern Mediterranean decreased by 30% from the period 2005-2009 ( $\sim 4200 \text{ t yr}^{-1}$ ) to the period 2010-2014 ( $\sim 3000 \text{ t yr}^{-1}$ ). This has been accompanied by a substantial decrease of the oldest (age class 4 and older) and youngest (class 0) individuals. Since 2010 the bottom trawler fleet was mainly targeting classes 1 and 2 and low, but persistent, removals of class 3 (Fig. 28).

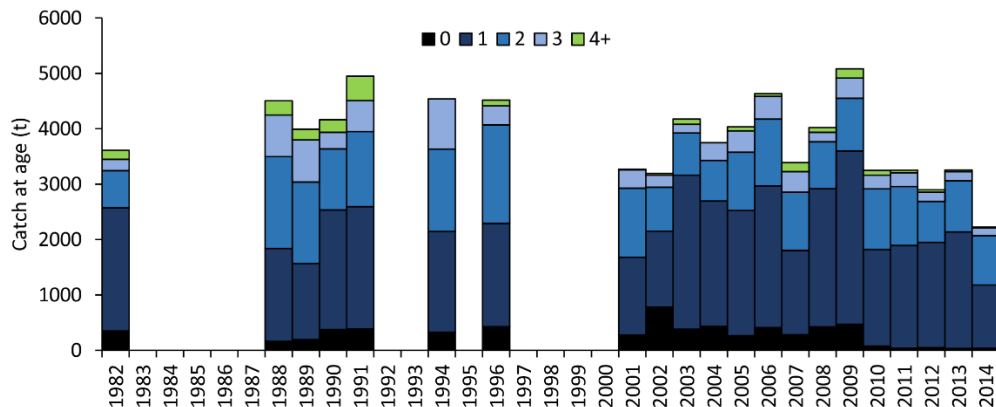


Fig. 28. Catch at age of hake caught by the bottom trawl fleet in Northwest Mediterranean Sea (data from STECF, 2015).

$F$ -at-age estimated by VIT followed the trend and held similar absolute values of those estimated by XSA (Fig. 29). Uncertainty in  $F_{ter}$  had higher impact on estimates of older ages. Estimates of  $F$  at age for ages 0, 1 and 2 by VIT were quite similar in seven of ten compared years, while  $F$  at age 3 showed larger differences regarding estimates derived from XSA. VIT produced closer values to XSA for the most fished ages (1 and 2), showing a relative error lower than 0.1 (Fig. 30a). Conversely, fishing mortality estimated by VIT in ages 0 and 3 produced larger relative errors ( $\sim 0.3$ ).

The official assessment performed by XSA considered ages 0-3 as the most fished ages ( $Fbar$ ). VIT was capable to identify the fishing mortality peak in 2006 as well as of tracking a rising trend of  $Fbar$  from 2007 to 2013 as XSA found (Fig. 31). However, VIT did not fit the absolute values of  $Fbar[0-3]$ , while the uncertainty derived from  $F_{ter}$  usually did not include the annual estimates obtained by XSA. Similar results were found when other  $Fbar$  were considered (Fig. 31). VIT failed to estimate the annual  $Fbar$  values, tending to underestimate the fishing mortality. Nevertheless, VIT could properly identify

that  $F_{bar}$  tended to increase between 2005 and 2013, as reflected by a relative error lower than 0.2 (Fig. 30b).

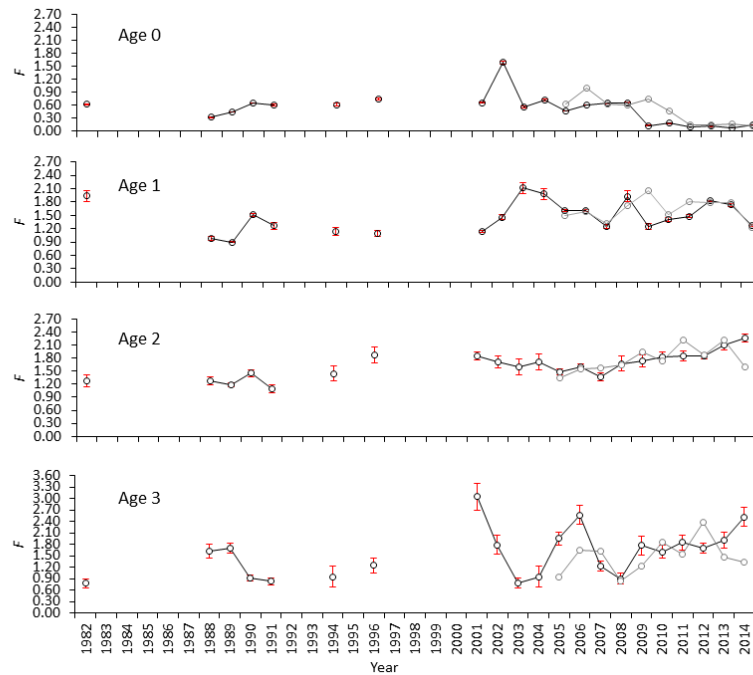


Fig. 29.  $F$ -at-age of hake produced by the bottom trawl fleet in Northwestern Mediterranean Sea. Estimations are provided by VIT (median, black) and XSA (as reported in official assessment, gray). A confidence interval of 95% derived from uncertainty in  $F_{ter}$  is shown (red bars).

$F_{bar}$  [1-3] estimated by VIT (mean= 1.60) was remarkably similar to that estimated by XSA (mean= 1.63). The  $F_{0.1}$  derived from the VIT analysis (mean= 0.31) was higher than estimated by the conventional VPA (mean= 0.26). Therefore, the overfishing level ( $F/F_{0.1}$ ) was slightly underestimated by VIT. Although relative error was equal or lower than 0.3 regardless ages computed for estimating  $F/F_{0.1}$ , more comparable values were obtained when age 0 was included (Fig. 30c).

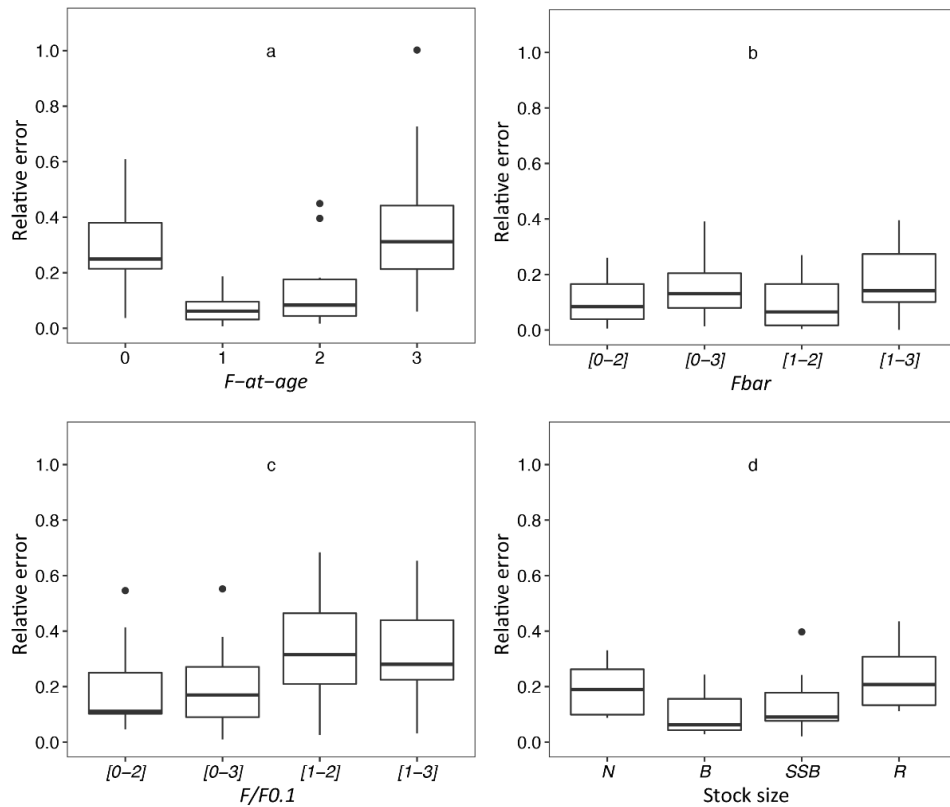


Fig. 30. Relative error in estimates of the stock indicators and reference points obtained by VIT (pseudo-cohort) and XSA (extended survivor analysis).

Trends in Stock ( $N$ ) and recruits ( $R$ ) numbers derived from by VIT assessment properly followed those estimated by XSA, although absolute values tended to be underestimated (Fig. 32). Uncertainty derived from  $F_{ter}$  was not translated into coincidence of two VPA estimates (under a confidence interval of 95%), meaning that equal parameterizing of two models did not lead fully comparable results.

Estimates of  $B$  and  $SSB$  by VIT were quite similar to those estimated by XSA, while the confidence interval always overlapped the data-rich VPA estimates. Despite the trend of the stock size showed remarkable agreement between methods, the data-limited method ranged relative errors of  $N$  and  $R$  around 0.2 (Fig. 30d). These deviations were propagated but exhibited lower variation over stock biomass and spawning stock biomass (median < 0.1).

## Summary of results chapter 2.1

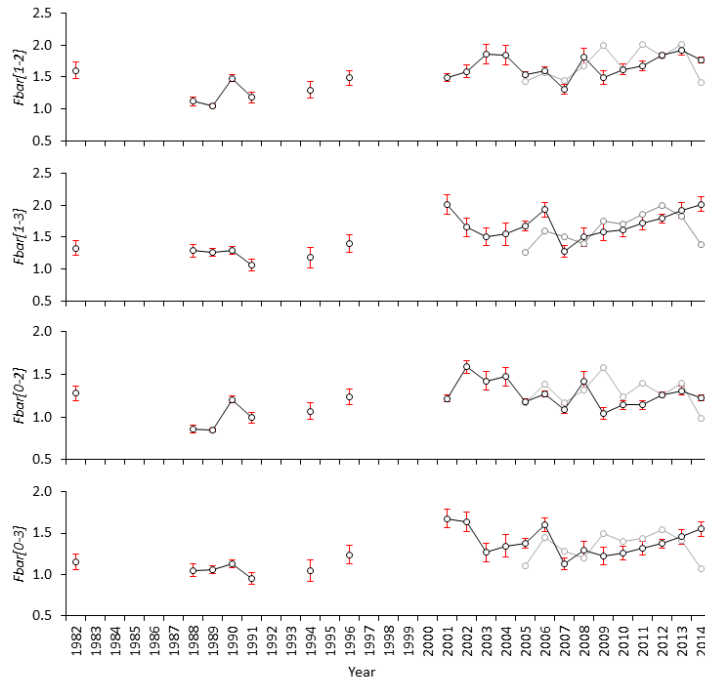


Fig. 31. Most fished ages ( $F_{bar}$ ) by the bottom trawl fleet in Northwestern Mediterranean Sea, estimated by VIT (median, black) and XSA (as reported in STECF 2015, gray). Confidence interval of 95% derived from uncertainty in  $F_{ter}$  is shown (red bars).

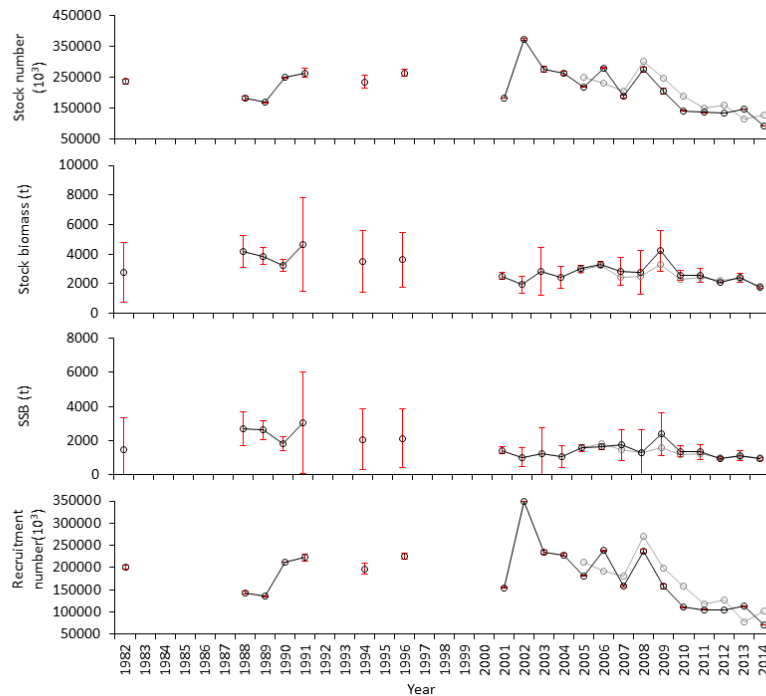


Fig. 32. Stock indicators of the European hake derived from the bottom trawl fleet catches in the Northwestern Mediterranean Sea. Estimations provided by VIT (median, black) and XSA (as reported STECF 2015, gray). Confidence interval of 95% derived from uncertainty in  $F_{ter}$  is shown (red bars).

### Exploring the historical indicators of hake stock

Fishing mortality has tended to remain constant since 1982. The most harvested ages were 1 ( $F \sim 1.3$ ) and 2 ( $F \sim 1.5$ ), while age 3 evidenced large annual variations (Fig. 29). According to findings obtained from comparison of the VIT and XSA outputs (2005-2014), we expect that fishing mortality of ages 1 and 2 from 1982 to 2001 had been able to provide better estimates than those obtained for ages 0 and 3. Accordingly,  $Fbar[1-2]$  showed an increased trend in fishing mortality from 1988 to 2013 (also showed by  $Fbar[1-3]$  and  $Fbar[0-3]$ ), suggesting that fishing mortality has not effectively decreased along four decades (Fig. 31). The stock size ( $N$  and  $R$ ) estimated by VIT may be underestimated around 20%. Therefore, the stock number and recruits in last years (2010-2014) may have declined over 52% and 58% regarding levels estimated in the eighties and nineties (Fig. 32).

### VIT performance under bias in life-history parameters

Estimates of  $SSB$  increased between 1% and 16% when  $k$  and  $M$  (positively correlated) were overestimated by 30%. When  $L_\infty$  was underestimated by 30% (negatively correlated with  $k$ )  $SSB$  decreased the same proportion (Table 12). Opposite variations of these parameters produced a reduction in estimates of  $SBB$  lower than 40%. These results indicated that under low uncertainty in estimates of the life-history parameters (<15%),  $SSB$  varied less than 20% compared to “known” value. However, uncertainty larger than 20% of  $L_\infty$ ,  $k$  and  $M$  produced high bias in estimate of  $SSB$  (> 38%).

Misspecification of life-history parameters (i.e. without considering the underlying relationship between parameters) may lead to large bias in stock size. Erroneously setting at least one of these three parameters usually produced large variation in the  $SBB$  estimates, even if this misspecification was low (<10%). For instance, an increase of the 20% in  $L_\infty$  resulted in an overestimation of 148% of  $SSB$  (Table 12).

## Summary of results chapter 2.1

Table 12. Impact of the uncertainty in estimates of asymptotic length ( $L_{\infty}$ ), constant growth rate ( $k$ ) and natural mortality ( $M$ ) on the relative value of spawning stock biomass of European hake in Mediterranean Sea in the pseudo-cohort analyses. Change of the  $SSB$  level is measured as increase (blue) or decrease (red) when the life-history parameters are above (>) or below (<) the “true” value. As the color is darker the  $SSB$  suffered larger changes compared to the “true” value.

Kind of bias	Parameter	variation	0.05	0.1	0.15	0.2	0.25	0.3
Uncertainty in estimates of life-history parameters	$k, M$	<,<	-0.04	-0.09	-0.14	-0.19	-0.24	-0.3
	$k, M$	>,>	0.04	0.07	0.1	0.12	0.14	0.16
	$L_{\infty}, k$	<,>	0.04	0.06	0.07	0.06	0.04	0.01
	$L_{\infty}, k$	>,<	-0.05	-0.11	-0.18	-0.25	-0.33	-0.41
	$L_{\infty}, k, M$	<,>,>	-0.11	-0.22	-0.33	-0.43	-0.52	-0.61
	$L_{\infty}, k, M$	>,<,<	0.11	0.22	0.32	0.41	0.49	0.55
wrong setting of life-history parameters	$M$	<	0.11	0.22	0.34	0.46	0.6	0.73
	$M$	>	-0.1	-0.2	-0.29	-0.38	-0.46	-0.54
	$L_{\infty}$	<	-0.29	-0.55	-0.78	-0.98	-1.16	-1.32
	$L_{\infty}$	>	0.32	0.67	1.06	1.48	1.94	2.44
	$k$	<	-0.21	-0.42	-0.61	-0.79	-0.96	-1.11
	$k$	>	0.23	0.46	0.71	0.97	1.24	1.51
	$k, M$	<,>	-0.17	-0.31	-0.43	-0.54	-0.63	-0.7
	$k, M$	>,<	0.19	0.41	0.66	0.94	1.26	1.61
	$L_{\infty}, k$	<,<	-0.24	-0.42	-0.57	-0.69	-0.78	-0.85
	$L_{\infty}, k$	>,>	0.29	0.64	1.06	1.57	2.17	2.87
	$L_{\infty}, k, M$	<,<,<	-0.18	-0.34	-0.47	-0.59	-0.68	-0.76
	$L_{\infty}, k, M$	<,>,<	0.02	0.02	0.01	-0.02	-0.06	-0.12
	$L_{\infty}, k, M$	>,<,>	-0.03	-0.08	-0.14	-0.2	-0.27	-0.35
	$L_{\infty}, k, M$	>,>,<	0.38	0.88	1.53	2.37	3.44	4.79
	$L_{\infty}, k, M$	<,<,>	-0.29	-0.5	-0.65	-0.77	-0.84	-0.9
$L_{\infty}, k, M$	>,>,>	0.2	0.43	0.68	0.95	1.25	1.57	

## Results chapter 2.2

Fig. 33 shows the fit to the length frequency data by year for the growth parameters estimated in this study. For this set of parameters,  $L_{50}$  varied around 11.5 cm TL and  $L_{95}$  around 14 cm TL (Fig. 34 left panel) while  $F/M$  decreased substantially from ca. 3.0 in 2007 to ca. 2.0 in the most recent years (Fig. 34 central panel), and the SPR was estimated to less than 0.1 but to have increased in the most recent four years (Fig. 34 right panel). The model outputs were similar for the other three life-history parameter sets in Table 13 (Figures S1, S2 and S3). The selectivity parameters remained quite constant (average  $L_{50} \sim 11.2$ ;  $L_{95} \sim 14.2$  across the four growth curves, Fig. 34, S1, S2 and S3), i.e. the values of the selectivity parameters were insensitive to the life-history ratios used as inputs to the LB-SPR model ( $CV < 0.5\%$  for both  $L_{50}$  and  $L_{95}$ , not shown).

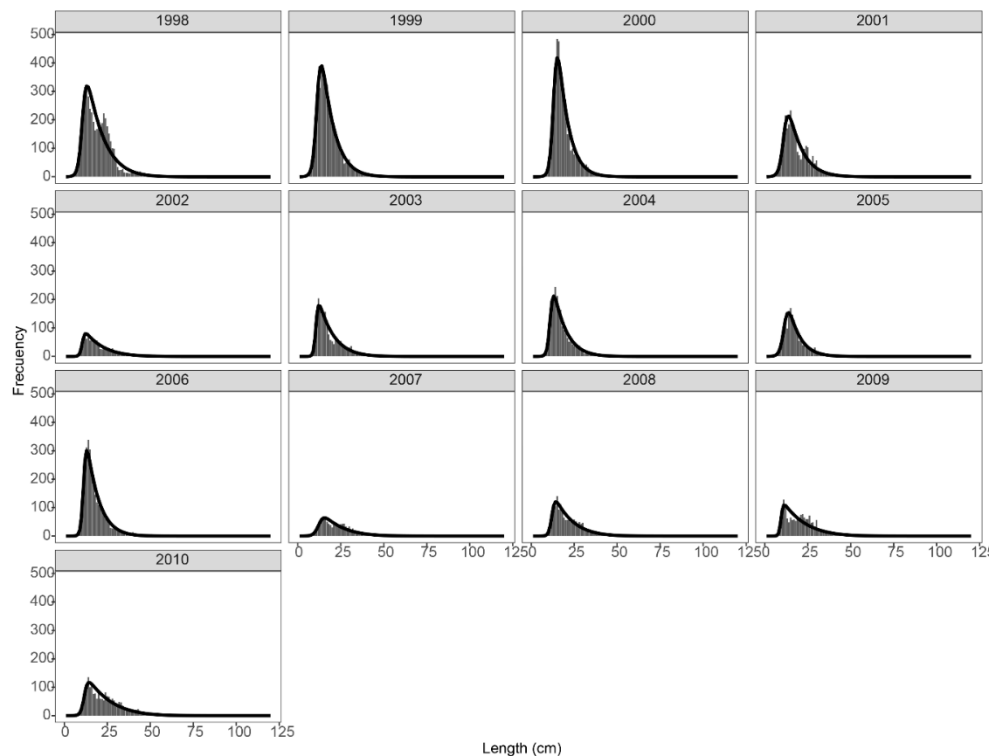


Fig. 33. Observed and LB-SPR model-predicted length-frequency histograms (bars and gray lines respectively) when the LB-SPR model was fitted using the life-history parameters derived from the Murcia region.

The highest ratio between fishing and natural mortality  $F/M$  (Fig. 35, Table S1) was for the lowest  $M/k$  value (2.34). Conversely, high  $M/k$  values (3.04 and 3.17), regardless of the growth hypothesis, led to the lowest estimates of  $F/M$ .

Table 13 von Bertalanffy growth parameters and natural mortality of hake in the Mediterranean Sea used in this study.  $M$  was estimated using PRODBIOM from the average  $M$  at ages 0-5 (Abella et al., 1997).

Source	Region	Sex	$L_{\infty}$ (cm)	$M$ ( $\text{yr}^{-1}$ )	$K$ ( $\text{yr}^{-1}$ )	Growth hypothesis	$M/k$
Mellon-Duval et al., 2010, model 3	Gulf of Lions	Male, female, undetermined	110	0.565	0.178	Fast	3.17
Mellon-Duval et al., 2010, model 1	Gulf of Lions	Female	100.7	0.62	0.236	Fast	2.63
Aldebert and Recasens, 1996	Catalan Sea	Female	100.7	0.29	0.124	Slow	2.34
Martínez-Bañós et al. ( <i>Data in Brief</i> )	Murcia	Combined	105.87	0.34	0.112	Slow	3.04

Although mean  $F/M$  derived from the four life-history parameters varied by ~79% per year from 1998 to 2010, the corresponding SPR estimates exhibited limited changes during some years (e.g., 1999, 2004 and 2006, Fig. 35, Table S1) or were clearly differentiated in others (e.g., 2007-2010, Fig. 35, Table S1). Fig. 33 indicates that the LB-SPR model was able to fit the length-frequency distributions for 1999, 2004 and 2006 better than those for other years, with the best fits coinciding with large samples sizes ( $n > 2,000$ ) (Table 14). Conversely, many of the annual models exhibiting poor fits to the length-frequency distributions (2007-2010, Fig. 33) coincided with sample sizes less than 2,000 individuals (Table 14). Thus, a good fit to the length frequency data by year usually led to similar SPR values (Fig. 35, Table S1), while poor fits tended to produce noticeable differences in the SPR estimates. We speculate that a larger number of length samples each year contributed to reducing the differences between the SPR estimates derived from the four life-history parameters.

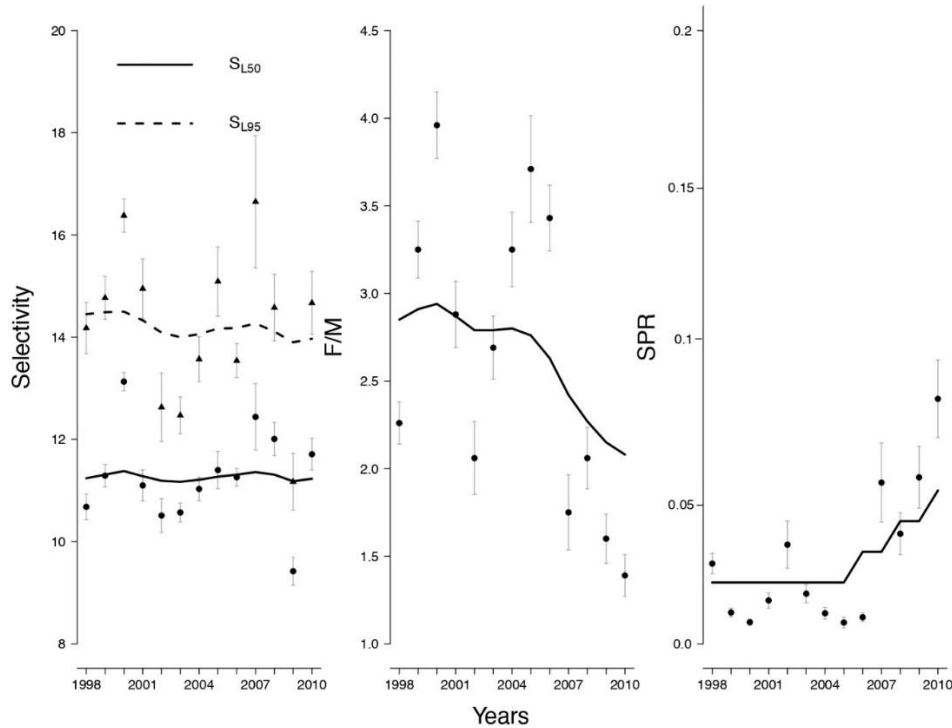


Fig. 34. LB-SPR outputs for the life-history parameters estimated for the Murcia region. Annual (individual symbol) and smoothed model (black line) estimates of selectivity,  $F/M$  and SPR. Confidence intervals (95%) are shown in vertical bars.

Poor fits for some years did not prevent the SPR estimates for the four sets of life-history parameters to be consistently below 5% from 1998 to 2006. Since 2007, the SPR tended to increase but was never higher than 8%. As for  $F/M$  estimation, the estimate of SPR did not depend on the growth hypothesis, but rather on  $M/k$ . The LB-SPR model was relatively insensitive to a CV of  $L_{\infty} < 0.15$ , but higher values caused differences in the SPR estimate that were greater than 10% (Fig. 36).

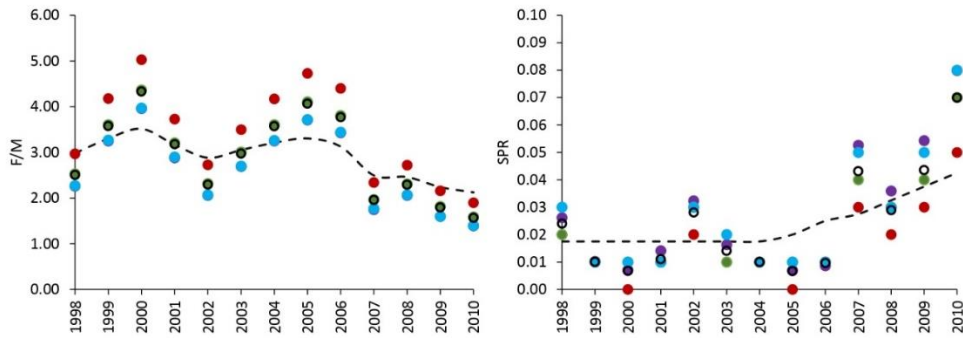


Fig. 35.  $F/M$  (left panel) and  $SPR$  (right panel) estimated using growth parameter estimated during this study (purple circles), by Aldebert and Recasens (1996) (red circles), from model 1 of Mellon-Duval et al.(2010) (green circles), and from model 3 of Mellon-Duval et al.(2010) (blue circles). Mean individual year (black empty circles) and mean smoothed model (dotted lines) estimates are shown.

The estimates of  $F/M$  and  $SPR$  were not very sensitive to the uncertainties in the life-history parameters, on year-to-year basis (Fig. 37a, b). The size frequency data grouped over three- or four-year periods substantially reduced the overall  $F/M$  ratio, which ranged from 1.5 to 4 (Fig. 37c). Thus, the relative fishing mortality estimated by the LB- $SPR$  model decreased by more than 42% when data were grouped. However, this reduction in  $F/M$  did not translate into changes in  $SPR$  estimates (Fig. 15d), and comparable  $SPR$  values were observed for annual (Fig. 37a) and year-grouped (Fig. 37d) assessments.

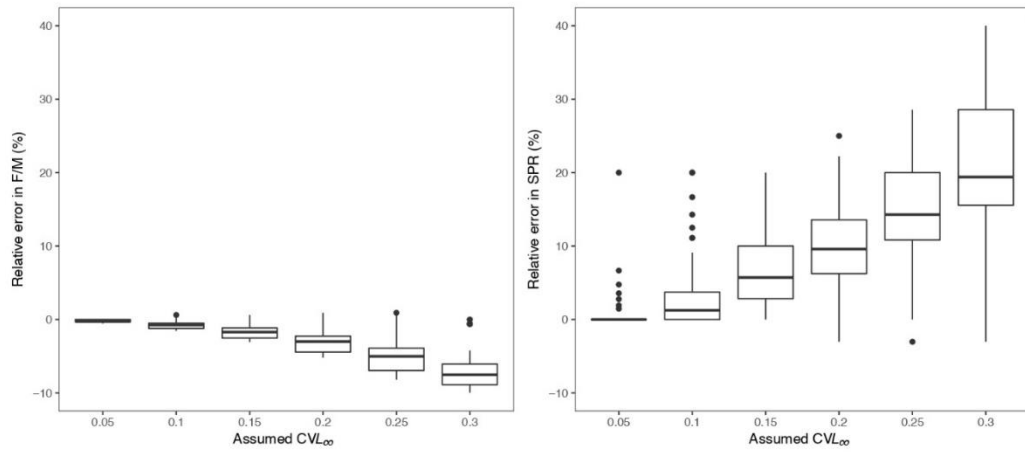


Fig. 36. Differences in estimates of  $F/M$  and  $SPR$  for various assumed values for the CV of  $L_{\infty}$  from the estimates when this CV is set to 0.001.

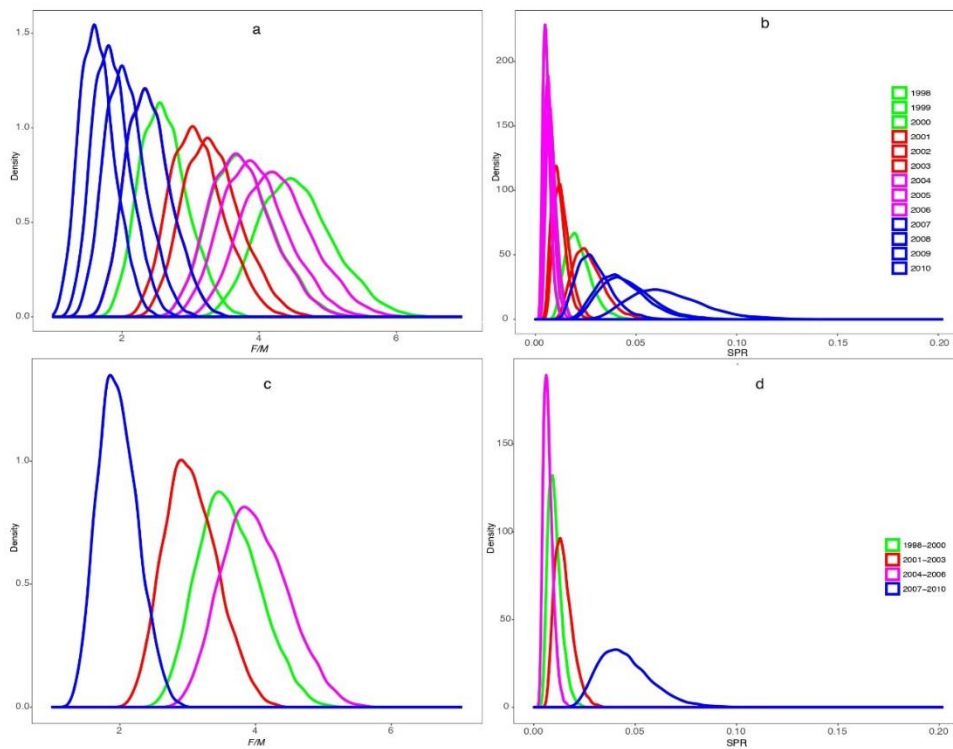


Fig. 37. Estimates of  $F/M$  (a,c) and  $SPR$  (b,d) for the European hake in the southwest Mediterranean. Results are shown by year (a,b) and when data are grouped (c,d).

Table 14. Landings (tonnes) and numbers of measured individual hake fished by trawlers during 1998-2010 in the Murcia region (SW Mediterranean, Spain).

Year	Landings (t)	Total individuals
1998	111.67	4436
1999	102.541	4505
2000	208.23	4199
2001	184.34	2697
2002	123.334	1041
2003	83.942	1990
2004	93.973	2231
2005	66.903	1691
2006	109.724	2985
2007	89.571	1416
2008	59.992	1608
2009	109.105	1594
2010	76.1	1948

### **Results chapter 2.3**

#### Catch reconstruction

From 2006 to 2014, the total reconstructed catch of lane snapper was estimated at 410 tons and the total catch of white grunt at 2,017 tons. The GOV data underestimated landings. The reconstructed landings of lane snapper and white grunt ranged from 3.25 to 8.52 times and 1.38 to 3.70 times greater than the official GOV landings (Fig. 38a). Most landings of lane snapper occurred in rural areas unmonitored by the GOV. In the case of white grunt, the official and reconstructed catches showed similar trends from 2006 to 2009. As of 2010, higher landings in rural areas led to the greater underestimations of government statistics (Fig. 38b). The GOV data suggested that the encircling gillnet fleet took the largest proportion of lane snapper (mean, 83%) and white grunt (mean, 99%). The reconstructed landings of lane snapper showed that the long-line fleet took the bulk of the catch during the first few years but that the catches have shifted to being dominated by encircling and bottom gillnets since 2011 (Fig. 18a,b). The reconstructed landings of white grunt increased the participation of bottom gillnets to 20% (Fig. 39c, d). On the other hand, the reduction of the average size of lane snapper remained hidden in the official GOV landings, and the reconstructed landings show the size to have decreased during the study period (Fig. 39). The average size of white grunt among the years remained stable in both monitoring programs.

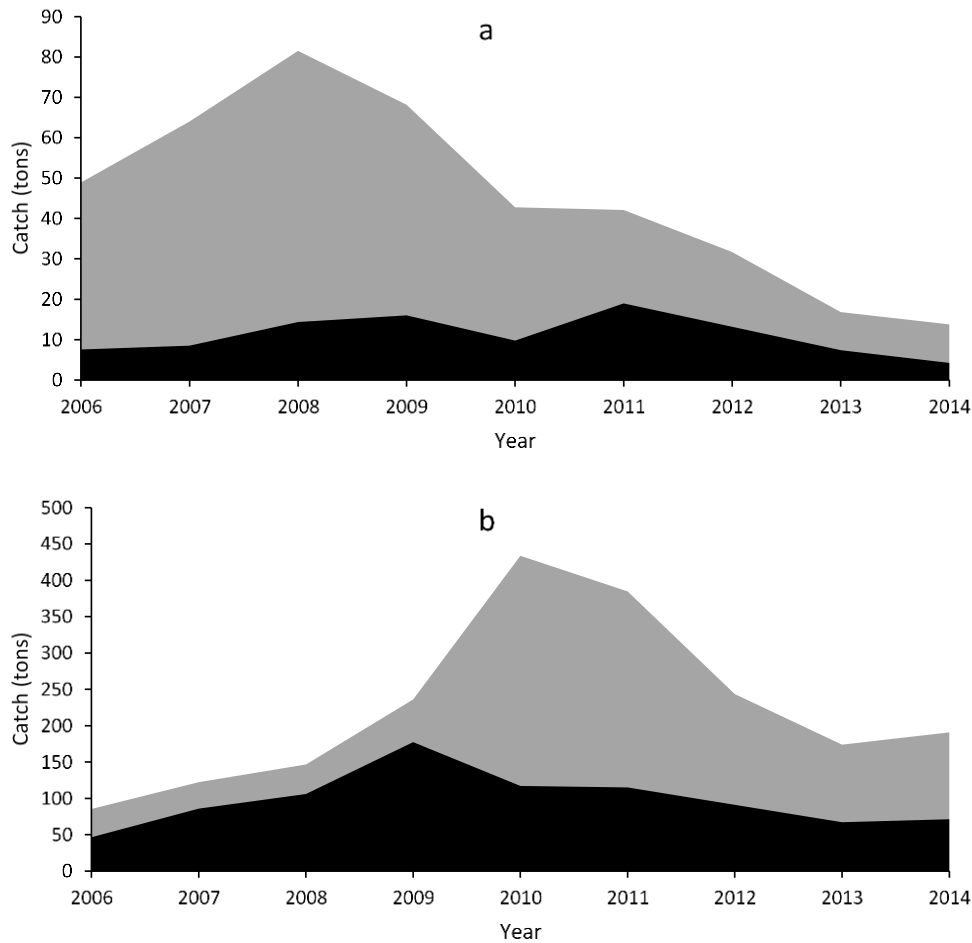


Fig. 38. Reported catch of lane snapper (a) and white grunt (b) by the National fishery agency involving urban areas (black), and total reconstructed catch (gray) employing both the rural landings by participatory-monitoring program and the urban catches by Government-monitoring program.

### Life-history parameters

The catches of lane snapper from the bottom gillnet and long-line fleets (from PAR) were mainly composed of sizes ranging from 14 to 40 and from 20 to 50 cm, respectively (see supplementary material, table S1). The encircling gillnet fleet (from GOV) mostly included sizes ranging from 19 to 41 cm (see supplementary material, table S2). Consequently, the  $vBGP$  calculated from the long-lines and bottom gillnets ( $L_{\infty}=64.94$  cm,  $k=0.19$  yr $^{-1}$ ,  $t_0=-1.58$  yr) produced a higher  $k$  than the  $vBGP$  calculated from the encircling gillnets ( $L_{\infty}=62.5$  cm,  $k=0.15$  yr $^{-1}$ ,  $t_0=-3.19$  yr). The length-based analysis for the white grunt population mainly included data from bottom gillnets (see supplementary material, table S3). Thus, the white grunt ranged from 15 to 36 cm, with a high  $k$  value ( $L_{\infty}=41.65$  cm,  $k=0.27$  yr $^{-1}$ ,  $t_0=-1.43$  yr).

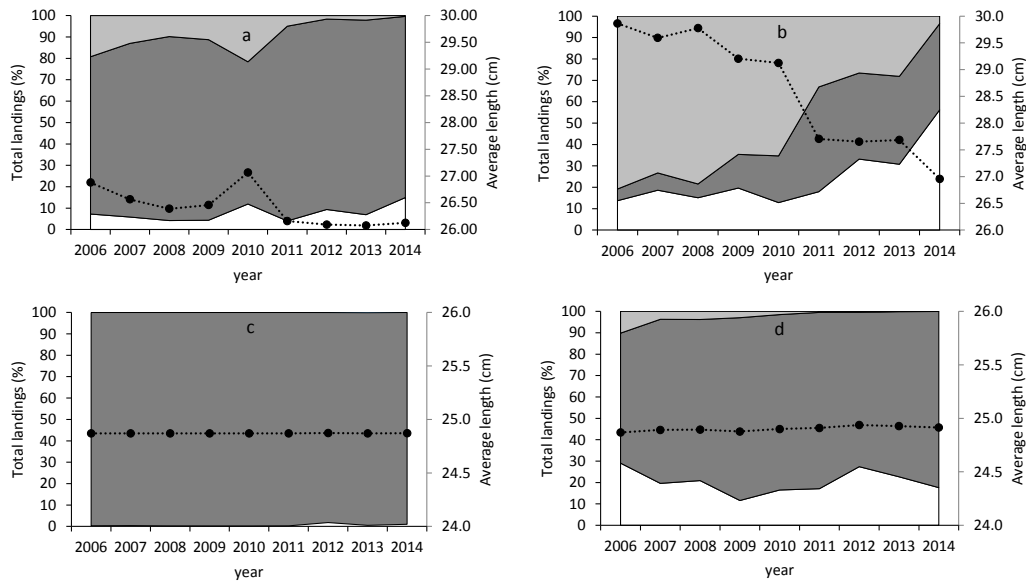


Fig. 39. Percentage of participation of longline (light grey), bottom gillnets (white) and encircling gillnets (dark grey) in the catches of lane snapper (a and b) and white grunt (c and d) from 2006 to 2014 on the left y axis. Average size of caught individuals on the right y axis (dotted line). Government-monitoring program (a and c) and Government-participatory combined data (b and d).

Natural mortality estimators based on life-history correlations tended to produce higher natural mortality rates in both species (Pauly's  $M=0.41-0.67 \text{ yr}^{-1}$  and Ralston's  $M=0.30-0.42 \text{ yr}^{-1}$ ). Similar results were produced by Gislason's *M-at-size* estimator (lane snapper average size,  $M=1.21-0.62 \text{ yr}^{-1}$ ; white grunt average size,  $M=1.97-0.31 \text{ yr}^{-1}$ ; see supplementary material, table S4). Additionally, the estimates from *M* estimators varied substantially depending on the  $L_{\infty}$  and  $k$  values used. Conversely, Hoenig's estimator based on  $T_{\text{max}}$  produced lower or moderate natural mortality rates (lane snapper,  $M=0.23 \text{ yr}^{-1}$  for  $t_{\text{max}}=19 \text{ yr}$ ; white grunt,  $M=0.15 \text{ yr}^{-1}$  for  $t_{\text{max}}=8 \text{ yr}$  and  $M=0.28 \text{ yr}^{-1}$  for  $t_{\text{max}}=18 \text{ yr}$ ). Abella's *M-at-age* estimator, which is also based on life-history correlations, produced lower mortality rates. The average of the *M* values for the first six ages ranged from 0.10 to 0.18  $\text{yr}^{-1}$  for lane snapper and from 0.12 to 0.20  $\text{yr}^{-1}$  for white grunt (see supplementary material table S5).

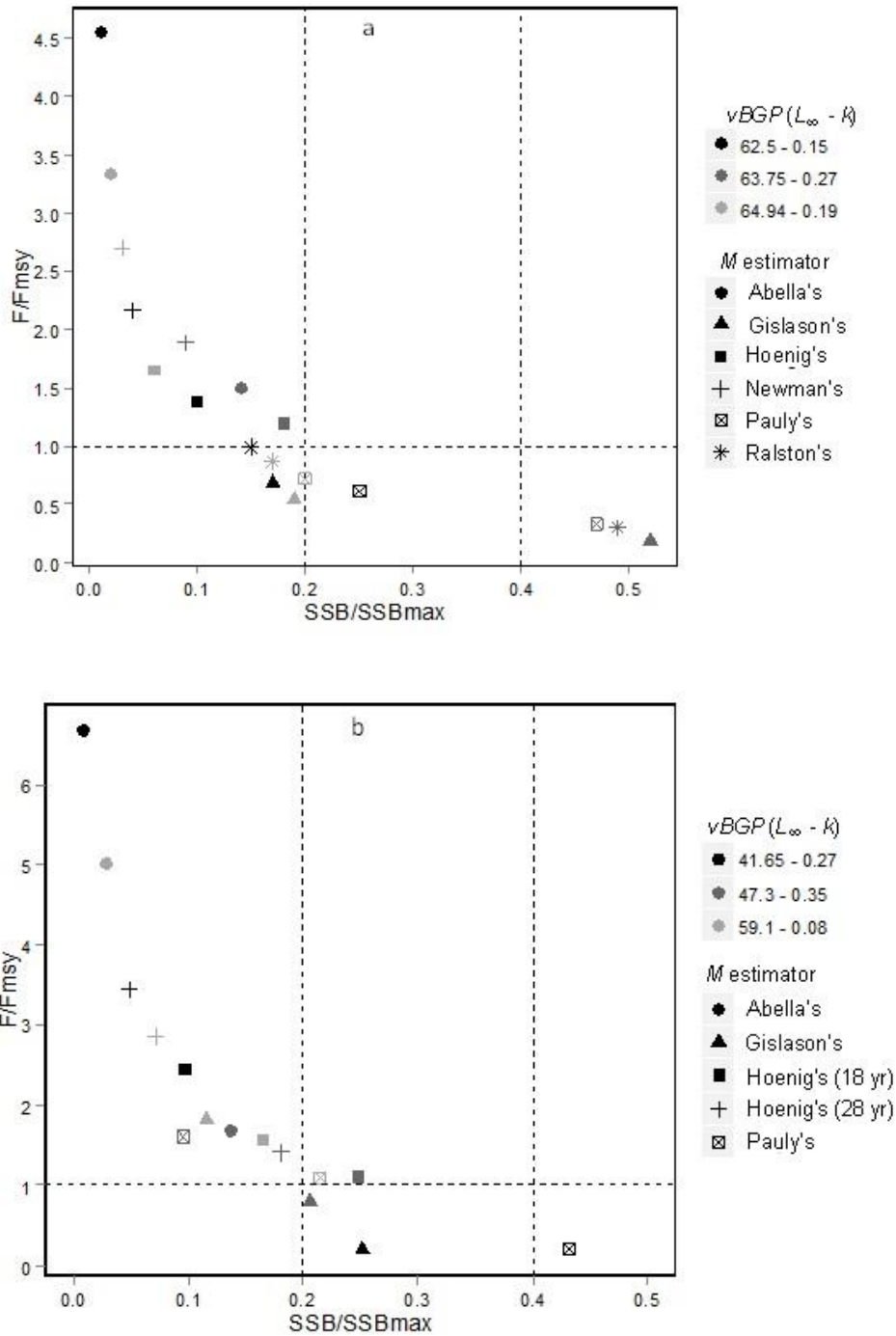


Fig. 40. Modified Kobe plot, involving three pairs of  $vBGP$  (symbols sharing the same color) and different natural mortality ( $M$ ) estimators (Different symbols). Life-history correlates for constant  $M$  such as the Pauly's (square and equis), Ralston's estimators (asterisk); the Gislason's  $M$ -at-size estimator (triangle); Abella's  $M$ -at-size estimator (circle);  $M$  based on maximum age ( $T_{max}$ ) by Hoening's (square) and  $M=0.13$  based on catch-curves from some lutjanids (plus). Hoening's estimator was performed to 19 years (lane snapper) and 18 and 28 year (white grunt) of maximum estimated age. Ralston's estimators were performed just for lane snapper, since these estimators are specific for lutjanids. Additional value from Hoening' estimator to white grunt is showed with plus symbol. Y/R analysis performed for populations of lane snapper (a) and white grunt (b) at the Central Guajira.

### Sensitivity tests

The population status of both species ranged from highly overexploited to underexploited (Fig. 40). This was mainly due to the natural mortality assumed in each stock assessment. The natural mortality estimated from Pauly's, Ralston's and Gislason's estimators yielded no overfished stocks ( $SSB/SSB_{max} > 40\%$ ) and overfished stocks ( $SSB/SSB_{max} < 20\%$ ), but overfishing was not occurring ( $F/F_{msy} < 1$ ). Conversely, stock assessments based on the  $M$  from Hoenig's and Abella's estimators indicated overexploitation. The exploitation levels of lane snapper and white grunt were also largely influenced by changes of  $vBGP$ . Abella's estimator showed the largest changes in  $F/F_{msy}$ . Gislason's, Pauly's and Ralston's estimators led to a large variation in  $SSB/SSB_{max}$ . Hoenig's estimator, which is based on the maximum estimated age, led to the lowest variation in exploitation level (Fig. 40).

### Population status

We chose three and two sets of  $vBGP$  to describe the growth of the lane snapper and white grunt, respectively. In the case of the lane snapper, the landings reduction and increased fishing mortality of small-sized fish during the study period suggested overfishing. Additionally, the high values of natural mortality contravene the low to moderate estimates of  $M$  obtained from unexploited snapper populations using catch-curve methods (Newman, 2002; Newman et al., 2000). For lane snapper, Frédoou et al. (2009) suggested low  $M$ . In performing a stock assessment of the lane snapper, we selected natural mortality rates of 0.13 and 0.24  $yr^{-1}$  from the Newman et al. (2000) and Hoenig's estimators. Further complications emerge for selecting  $M$  for the white grunt, as larger gaps in its estimation remain when indirect estimators are used. Therefore, we selected the natural mortality rates of 0.15 and 0.24  $yr^{-1}$  by focusing on methods that use the maximum estimated age, as suggested by Then et al. (2015), and avoided using  $M$  values higher than 0.35  $yr^{-1}$ , as recommended by Potts (2000).

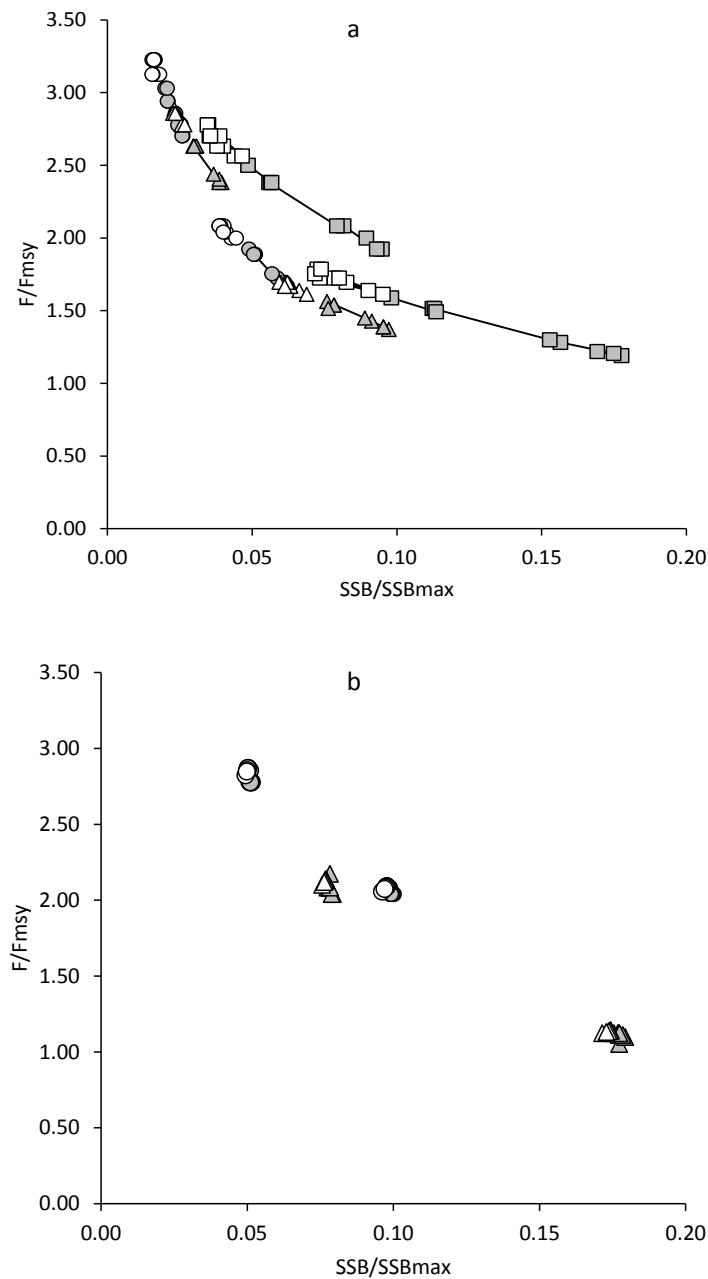


Fig. 41. Precautionary plot, establishing the relative threshold reference point (SSB20%) and F ratio (F/F<sub>msy</sub>). Historic analysis performed from Government (empty symbols) and reconstructed landings (gray symbols). Late years show higher overexploited level. Lane snapper (a) when  $L_{\infty}=64.94$ ,  $k=0.19$ ,  $M=0.23$  (lowermost circles) and  $M=0.13$  (upper circles),  $L_{\infty}=63.75$ ,  $k=0.27$ ,  $M=0.23$  (lowermost squares) and  $M=0.13$  (upper squares) and  $L_{\infty}=63.75$ ,  $k=0.27$ ,  $M=0.23$  (lowermost triangles) and  $M=0.13$  (upper triangles). White grunt (b) when  $L_{\infty}=41.65$ ,  $k=0.26$ ,  $M=0.24$  (lowermost circles) and  $M=0.15$  (upper circles) and  $L_{\infty}=59.01$ ,  $k=0.08$ ,  $M=0.24$  (lowermost triangles) and  $M=0.15$  (upper triangles).

The lane snapper was estimated to be more overexploited when the assessment was based on landings from the GOV (Fig. 41a). Underestimation of long-line catches in urban areas meant that large fish sizes were unreported. Furthermore, the stock

assessment performed using the GOV data led to less change between earlier and later years than that obtained when the PAR data were also included in the assessment. This means that a worsening in the overexploitation status of the lane snapper was smoothed because the GOV did not perceive the increase in fishing mortality of smaller-sized fish.

The status of the white grunt population was insensitive to whether government and unified government-participatory monitoring data were analyzed (Fig. 41b). The encircling gillnet fleet showed a large participation in urban and rural landing points, reporting between 61% and 83% of the total reconstructed landings. Additionally, the encircling gillnet fleet caught fish sizes similar to the those for the bottom gillnet fleet, implying that between 89% and 99% of landings involved a similar length structure. These conditions contributed to an unchanged population status between the GOV and reconstructed data. Therefore, the total biomass harvested by the fleet should be a key aspect to consider in the management of the white grunt population.

#### Management simulations

The lane snapper and white grunt populations are estimated to have been overexploited since 2006. The dominance of the encircling gillnet fleet in lane snapper and white grunt catches, which mainly removed small-sized fish, placed the relative spawning stock biomass (SSB/SSB<sub>max</sub>) of both species below 20%. The trends suggest higher overfishing between the earlier and later years studied (Fig. 41). The simulation outputs covered the selected  $vBGP$  and  $M$ , as well as the two different recruitment models (Constant and B&H). The B&H recruitment model allowed less reduction of  $\phi$  than the constant model for both species, regardless of which set of  $vBGP$  was used. The management scenarios “people first” ( $1 \geq \phi > 0.82$ ) and “only harvesting of medium sizes” ( $\phi = 1$ ) led to the lowest reduction of the bottom gillnet fleet effort (Table 16). From a social point of view, this is desirable because  $Y/R$  increased over initial simulation levels (see scenarios 1 and 3, Fig. 42). Furthermore, many fishers who inhabit rural areas would be able to secure income and food (Table 15). Conversely, if priority is given to obtaining the highest of either income or catch, the encircling gillnet fleet should be maintained (Fig. 42 and Fig. 43). This would, however, lead to a reduction in the effort of most

bottom gillnet vessels (“the best catches”;  $0.28 \geq \phi \geq 0$ ) and long-line vessels (“the volume does matter”;  $0.47 \geq \phi \geq 0.2$ ). This means that rural communities would be seriously affected (Table 15 and Table 16), and the recovery of the lane snapper population would be both lower and slower than in previous scenarios (Fig. 42). Focusing the fishing effort on the highly valued large fish sizes increases long-line yields. This would require an important reduction in the effort of the bottom and encircling gillnet fleets ( $0.63 \geq \phi \geq 0.40$ ), which would be socially unsustainable.

The white grunt fishery has a marginal long-line harvest (lower than 5% of total catch), so it was not necessary to consider long-lines in the management scenarios. The “people first” scenario, which prioritized the bottom gillnet fleet, also allowed some participation of the encircling gillnet fleet ( $0.46 \geq \phi \geq 0.06$ ). This would prioritize the protection of employment and food security for rural and urban fishers (scenarios 1 and 2; Fig. 43). Conversely, prioritizing the largest catches demands a reduction in the effort of the encircling gillnet fleet ( $0.76 \geq \phi \geq 0.4$ ) and the total elimination of the bottom gillnet fleet ( $\phi = 0$ ), a total reduction of 66.3% to 86.5% of the operative fleet effort (Table 16). Finally, a homogeneous reduction of all fleet efforts for both species would not offer higher yields but would result in greater negative impacts on fishery incomes and food security (Fig. 42 and Fig. 43)

When scenarios that favored the traditional bottom gillnet fleet (“people first” and “just harvesting of medium sizes”) were optimized, the fishing effort on lane snapper ( $1.62 \geq \phi \geq 1.1$ ) and white grunt ( $2.6 \geq \phi \geq 1.19$ ) could be increased. Optimization of a more dominant fleet, the encircling gillnets (“the volume does matter”), allowed the fishing effort on lane snapper to be increased ( $1.75 \geq \phi \geq 1.27$ ). However, this optimization would require harvesting of white grunt to be reduced to achieve SSB20% ( $\phi = 0$ ) (Table 16), suggesting that joint management of both species depends on the fishing effort of the encircling gillnets on the white grunt population.

<i>Lutjanus synagris</i> management scenarios		Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher and high-valued catches and ensuring food security in the rural communities (long-line and bottom gillnet) Optimizing both highly valued sizes (long-line) and
2	The "best" catches	Removing gear that fishes the highest proportion of medium sizes (bottom gillnet)	Eliminating the fleet formed by the most vessels from rural communities (bottom gillnet)	the highest catches (encircling gillnet), even if food security in rural areas could be affected (bottom gillnet)
3	Only harvesting medium sizes	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and small sizes (encircling gillnet)	Keeping the fleet formed by the most vessels from rural communities (bottom gillnet)	Encouraging both the cheapest fishing trips and food security, especially in rural areas
4	The volume does matter	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and medium sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
5	Targeting the highest-valued sizes	Reducing as much as possible catches with a high proportion of both small sizes (encircling gillnet) and medium sizes (bottom gillnet)	Keeping the fleet that underwent the greatest fishing reduction during the study period	Promoting the highest-valued as well as the lowest catches. Food security of rural communities is poorly considered
6	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)
<i>Haemulon plumieri</i>		Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher catches and ensuring food security in the rural communities (bottom gillnet)
2	The volume does matter	Removing gear that fishes the highest proportion of medium and large sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
3	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)

Table 15. Management scenarios of lane snapper and white grunt fisheries according to fish size, deployed fleet and food security and incomes of fishing communities. Management scenarios are separated by species given that the fleets act differently on sizes and total catches. Management scenarios were incorporated into simulations by controlling the caught sizes by gear (Fish size column) and fleets in the fishery (Deployed fleet column). The third column (Food security and profits of fishing communities) shows the effect caused by controlling for fish size and fleet, but it is not directly incorporated into the simulation model.

## Summary results chapter 2.3

Species	Scenario	$vBGP$ and $M$	Recruit model	Management measures ( $\phi$ by gear)			Optimization of $\phi$	
				Bottom gillnet	Encircling gillnet	Long-line		
<i>L. synagris</i> (Lane snapper)	People first	a	Constant	0.82	0	0.82	N/A	
			B&H	1	0.15	1	1.1	
		b		Constant	0.9	0	0.9	N/A
				B&H	1	0.27	1	1.2
	The "best" catches	a		Constant	0	0.92	0.92	N/A
				B&H	0.25	1	1	1.2
		b		Constant	0	0.9	0.9	N/A
				B&H	0.28	1	1	1.2
	Only harvesting medium sizes	a		Constant	1	0.12	0.12	1.15
				B&H	1	0.4	0.4	1.5
		b		Constant	1	0.16	0.16	1.2
				B&H	1	0.45	0.45	1.62
	The volume does matter	a		Constant	0.21	1	0.21	1.35
				B&H	0.45	1	0.45	1.75
		b		Constant	0.2	1	0.2	1.27
				B&H	0.47	1	0.47	1.65
	Targeting the highest- valued sizes	a		Constant	0.4	0.4	1	2.9
				B&H	0.6	0.6	1	3.9
		b		Constant	0.42	0.42	1	3.35
				B&H	0.63	0.63	1	4.7
Lower fishing	a		Constant	0.51	0.51	0.51	N/A	
			B&H	0.65	0.65	0.65	N/A	
	b		Constant	0.52	0.52	0.52	N/A	
			B&H	0.69	0.69	0.69	N/A	
<i>H. plumierii</i> (white grunt)	People first	c	Constant	1	0.06	1	1.19	
			B&H	1	0.22	1	1.7	
		d		Constant	1	0.29	1	1.9
				B&H	1	0.46	1	2.6
	The volume does matter	c		Constant	0	0.4	1	N/A
				B&H	0	0.55	1	N/A
		d		Constant	0	0.6	1	N/A
				B&H	0	0.76	1	N/A
	Only harvesting medium sizes	c		Constant	0.3	0.3	1	N/A
				B&H	0.4	0.4	1	N/A
		d		Constant	0.46	0.46	1	N/A
				B&H	0.59	0.59	1	N/A

Table 16. Simulations of management scenarios, based on.  $L_{\infty}=64.94$  cm,  $k=0.19$  yr<sup>-1</sup>,  $t_0=-1.58$ ,  $M=0.23$  yr<sup>-1</sup> (a) and  $L_{\infty}=63.75$ ,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-0.66$ yr,  $M=0.13$  yr<sup>-1</sup> (b) to lane snapper and  $L_{\infty}=41.65$ cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0=-1.43$ yr,  $M= 0.15$  yr<sup>-1</sup> (c) and  $L_{\infty}=41.65$ cm,  $k=0.27$  yr<sup>-1</sup>,  $t_0 =-1.43$ yr,  $M=0.24$  yr<sup>-1</sup> (d) for white grunt populations. Each scenario produced four results according to  $vBGP$ - $M$  and either the constant or Beverton and Holt (B&H) recruitment models. Simulation cases in which the prioritized fleet could maintain a fishing multiplier ( $\phi$ ) above 1 were maximized, and other gears were eliminated (optimization column).

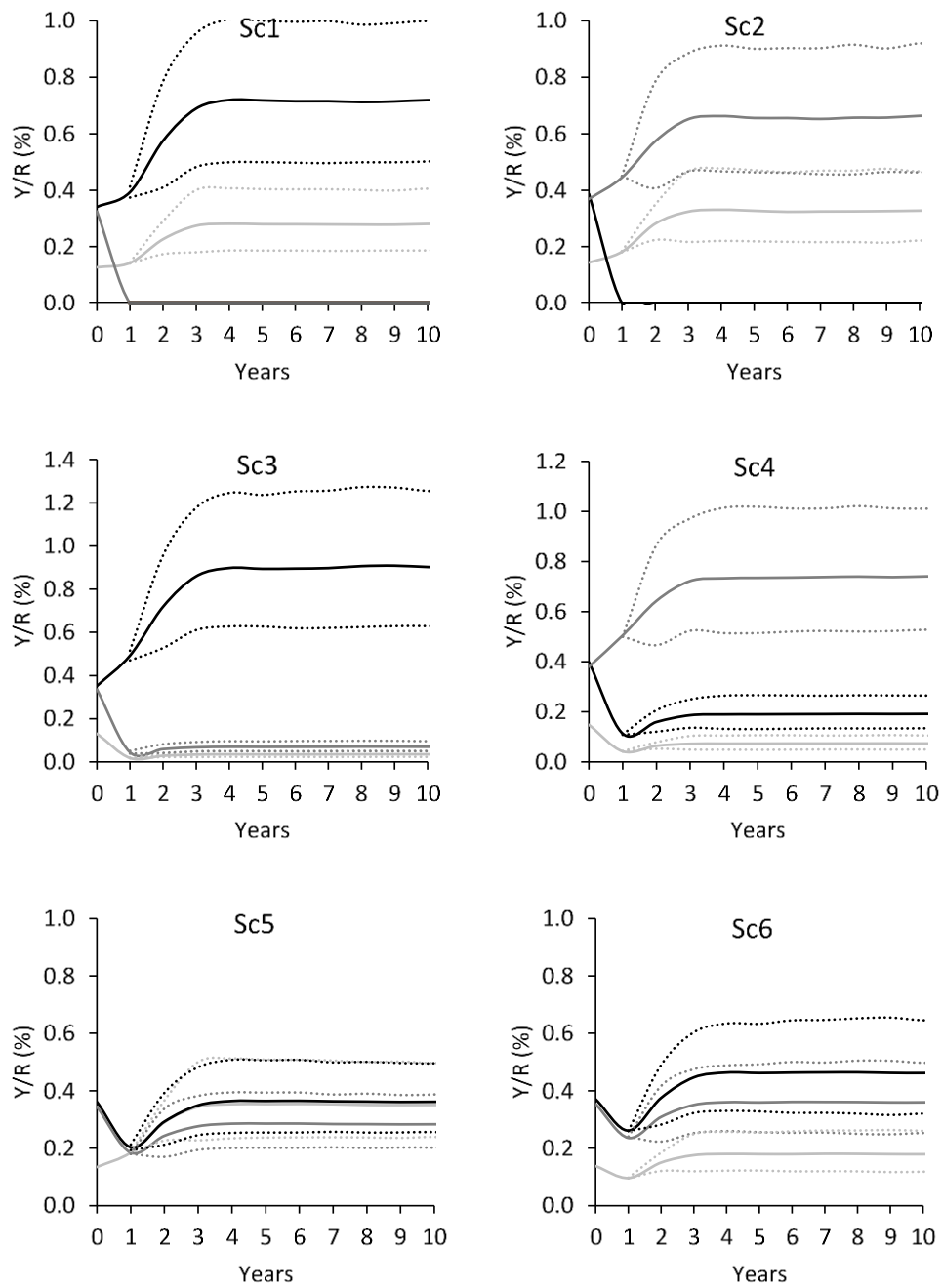


Fig. 42. Simulation of lane snapper yield per recruit (Y/R) by longline (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets, showing the mean and 95% confidence intervals of 1000 iterations (dotted line). Scenarios (Sc) are performed according to management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment models for the parameter set defined as  $L_{\infty}=64.94$ ,  $k=0.19$  and  $M=0.23$ . All scenarios accomplished reaching a relative SSB20% in a period of 8 years.

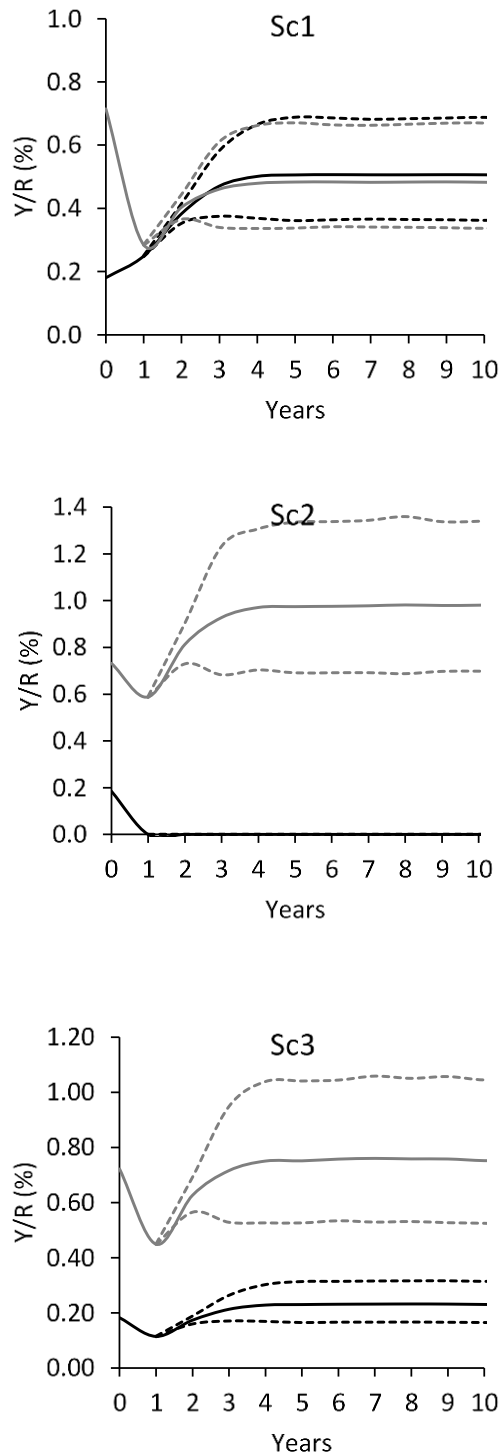


Fig. 43. Simulation of white grunt yield per recruit (Y/R) by longline (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets, showing the mean and 95% confidence intervals of 1000 iterations (dotted line). Scenarios (Sc) are performed according to management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment models for the parameter set defined as  $L_{\infty}=41.65$ ,  $k=0.27$  and  $M=0.15$ . All scenarios accomplished reaching a relative SSB20% in a period of 8 years.

**Results chapter 2.4**

## Life history values

The number of sampled individuals in 2012 ( $n=498$ ), 2013 ( $n=663$ ), 2014 ( $n=1021$ ) and 2015 ( $n=572$ ) provided sufficient sample size to fit the LB-SPR model (Fig. 44). The median size of the Peruvian grunt caught by the beach artisanal fishers ranged from 26.5 cm to 28.5 cm. Size at first maturity for male and females were comparable ( $L_{50} \sim 21$  cm) (Figure S1), but  $L_{95}$  was reached later by males (41.21 cm) than by females (35.87 cm). Overall, this fishery targets mature individuals with no decrease in selectivity, so likely fulfills the LB-SPR assumption of asymptotic selectivity

Given that SPR is defined by the age truncation of the stock (here observable from maximum caught sizes relative to asymptotic length), defining  $L_{\infty}$  can be a highly influential parameter in the estimate of LB-SPR. Length-based estimates of  $L_{\infty}$  were larger than 58 cm, while those estimates based on FishLife ranged from 50.45 cm (based on borrowed Burrito grunt values) to 62.05 cm (updated with the Peruvian grunt length-based estimates) (Table 11). The FishLife results suggested that the absence of specific information on life-history parameters for Peruvian grunt in FishBase may inject bias when using those parameters in a model.

Natural mortality largely varied depending on which estimator was considered (Table 11). Values derived from the  $L_{\infty}$  and  $k$  parameters ranged from 0.223 to 0.563.  $M/k$  ratios ranged from (1.63-2.02), substantially different from assumption that  $M/k$  is approximately 1.5 (Froese et al., 2018a, 2018b).

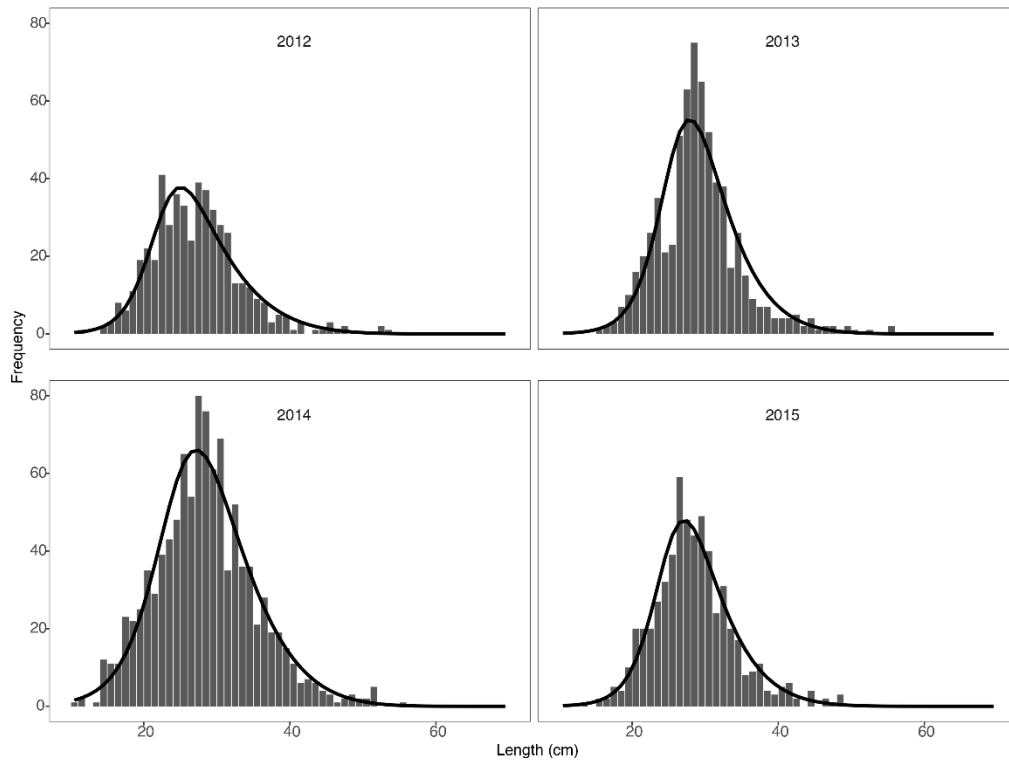


Fig. 44. Length-frequency structure of the Peruvian grunt stock fitted (black line) by the LB-SPR model.

#### SPR estimates

The SPR distribution obtained under the assumption of proportional contribution of the 118  $L_{\infty}$ - $M/k$  combinations produced two main peaks around 0.10 and 0.25 for the four years (Fig. 45a). Both modes indicate the Peruvian grunt stock is below the target (SPR < 0.4, Myers et al. 1994), but is not clear whether the stock was below or above the limit reference point of  $SPR_{0.2}$ . The attribute of multiple modes was common in these results (Fig. 46, S2, S3 and S4), and values higher than 0.4 were linked to the Burrito grunt life-history parameters obtain from FishLife.

When life-history parameters were adjusted to triangular distribution (Fig. 47), the SPR distribution acquired a unimodal shape (Fig. 45b and c). Regardless which criteria was used to define the minimum and maximum distribution values, the SPR distribution tended to be below 0.2. The overall distribution of SPR values was expectedly more precise when imposing the more limited minimum and maximum  $L_{\infty}$  and  $M/k$  values.

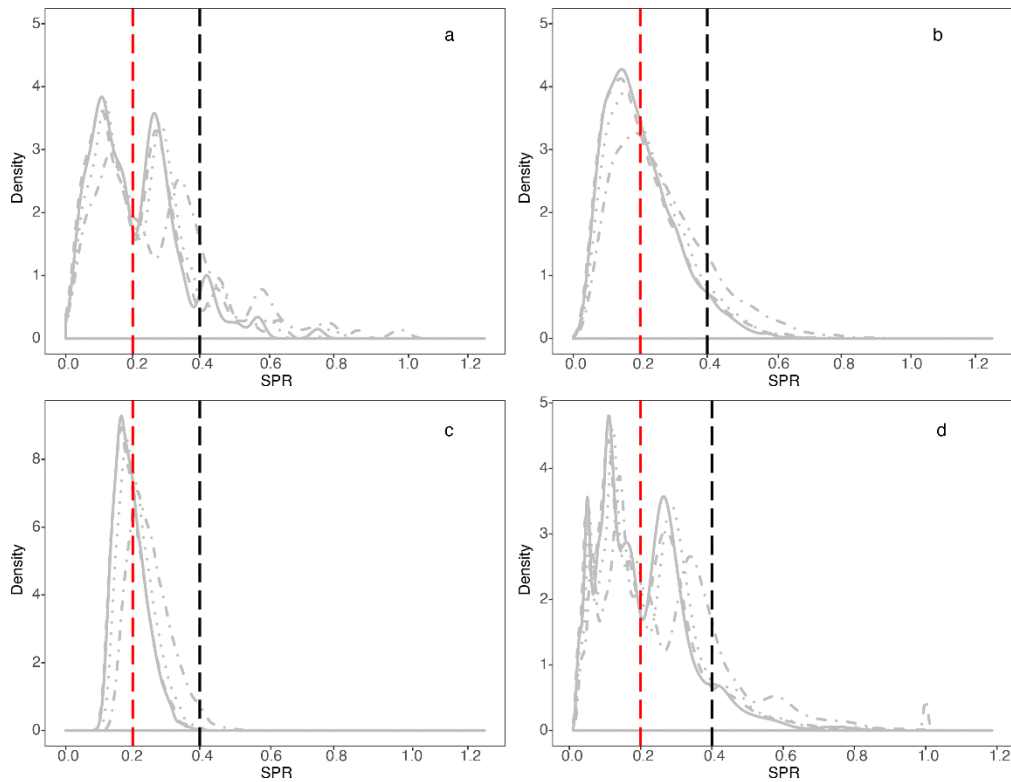


Fig. 45. SPR distribution derived from a) equal probability contribution of all life-history parameter sets, b) triangular distribution of life-history parameters, c) triangular distribution after constraining the minimum and maximum values of  $L_{\infty}$  and  $M/k$  to the first and third quartiles and d) Monte Carlo simulation. The LB-SPR analysis was performed in 2012 (dashed line), 2013 (dotted line), 2014 (dotdash line) and 2015 (solid line). Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

Using 10,000 draws of the  $L_{\infty}$ - $M/k$  pairings also led to  $SPR < 0.4$  (Fig. 45d). However, this analysis revealed three SPR peaks below  $SPR_{0.2}$ . The SPR distribution also indicated very low probability that the Peruvian grunt stock SPR is  $> SPR_{0.4}$ . Low probability above the target SPR value again derived from the Burrito grunt life-history parameters.

The updated routine of FishLife, which estimated the life-history parameters from both global (FishBase) and local (length-based) data, produced the most consistent SPR distributions (Fig. 48, "Peruvian grunt-updated" panel).

The SPR distribution was also influenced by the contribution of large individuals in the perceived stock structure. The SPR distribution in 2014 (Fig. 48), which reported larger individuals than other years, was more scattered than in 2012 (Fig. S5), 2013 (Fig. S6) and 2015 (Fig. S7).

## Summary results chapter 2.4

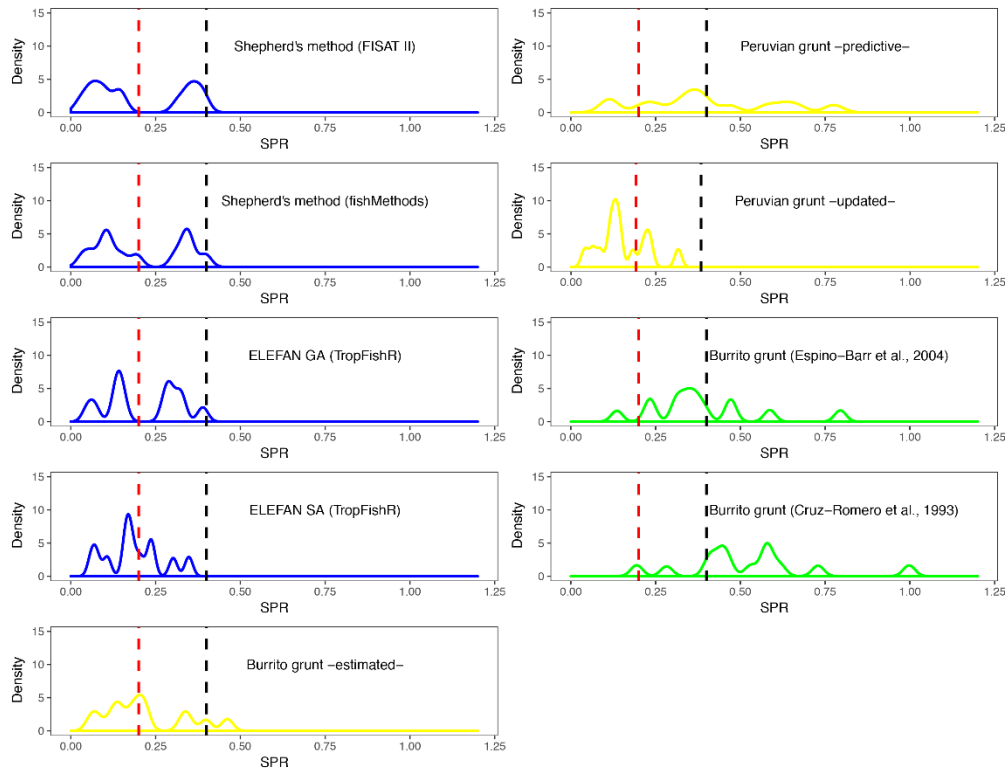


Fig. 46. SPR distribution in 2014 produced by proportional contribution of the life-history parameters derived from the nine different methods to estimate the  $vBGP$ . Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

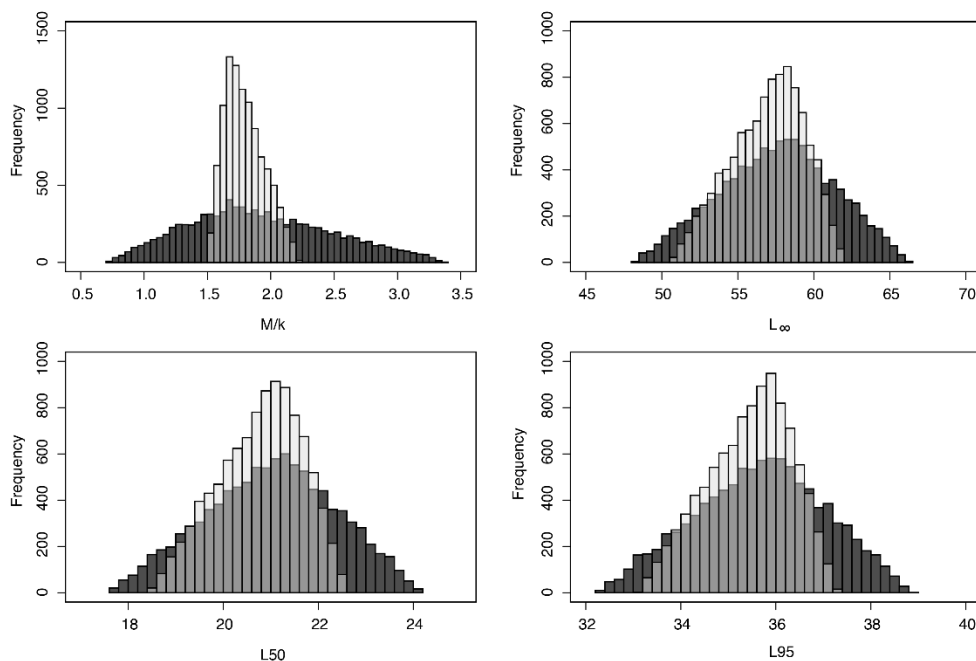


Fig. 47. Uncertainty in life-history parameters incorporated into  $SPR$  estimates through triangular distribution. Distribution values defined by the maximum, minimum and median values estimated for each parameter (dark grey bars). Distribution values of the minimum and maximum values of  $L_{\infty}$  and  $M/k$  constrained to the first and third quartiles (light grey bars).

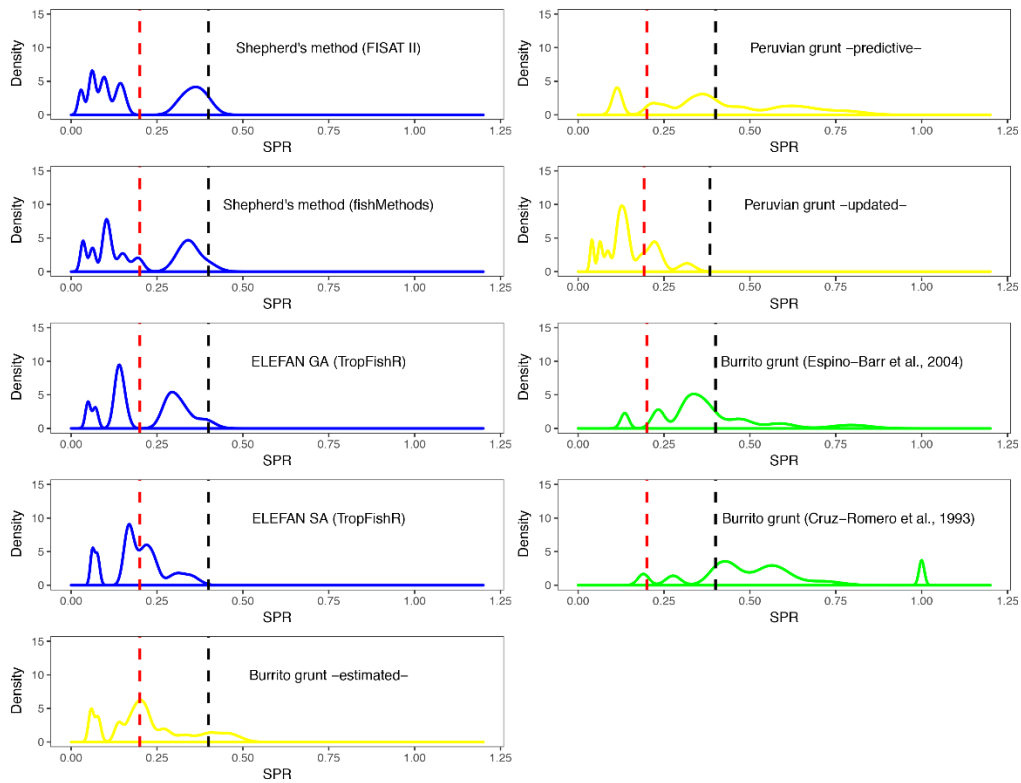


Fig. 48. SPR distribution in 2014 obtained from the life-history parameters distribution uncertainty estimated by the Monte Carlo simulation. Length-based (blue lines) and FishLife (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.



### 3. DISCUSSION



This thesis was aimed at addressing the reliability of the data-poor stock assessment methods to provide advice on the status of small-scale fisheries lacking catch history. The thesis findings indicate that stock assessment methods based on steady-state assumptions may contribute to define the stock status of the bulk of the non-assesd world’s small-scale fisheries. However, the stock assessment of these fisheries is often surrounded by an unquantified level of input data biases. Thus, the stock assessment process must deal with the uncertainty produced by the model outputs.

Box 4. Anchoring the assessment of small-scale fisheries

A simile between a “ballestilla” (a kind of handline used by artisanal fishers in the Caribbean and Ocean Pacific to catch demersal fish on narrow rock bottoms) and the approach used in this thesis for defining the status of small-scale fisheries.

The surface balloons represent the perceived status of small-scale fisheries under the data availability and assumptions of the assessment models used. The underwater balloon represents the availability of only isolated or sparse data (no catch history is available). Each of the four hooks represent the sequential case studies used in this thesis to determine the challenges, opportunities and implementation of good practices to assess small-scale fisheries.

The four case studies help to determine the anchoring aspects that holds a proper assessment of small-scale fisheries. New information will be included in the figure, in accordance with the development of the thesis discussion chapter.

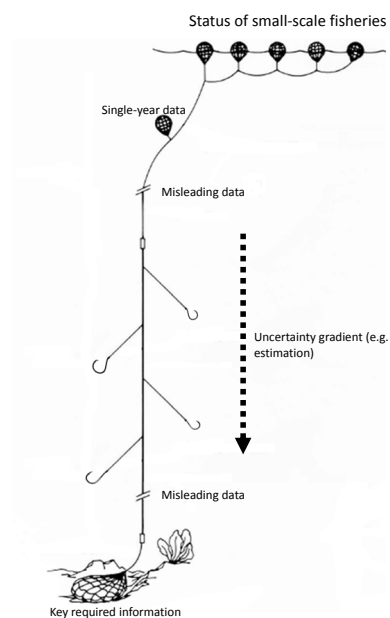


Figure of “ballestilla” is modified from Arana and Vega (2000).

## 3.1. Performance of data-limited stock assessment methods

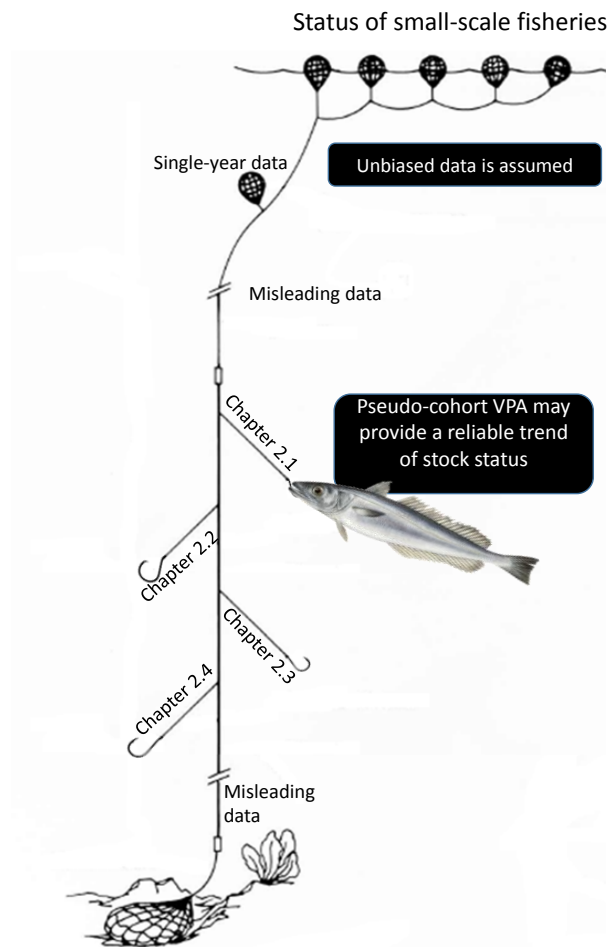


Fig. 49. Assuming unbiased input data to assess a stock by a pseudo-cohort VPA. Figure of "ballestilla" is modified from Arana and Vega (2000).

Two main ways are used by the fishery science to determine the performance of data-limited stock assessment models. Simulated data of no-real populations (so-called "silicone fish populations") (Beddington and Kirkwood 2005) are used to determine the performance of these assessment models. This way is useful in the sense that many population structures, several life-history traits and different quality data may be tested (Hordyk *et al.* 2015b,a; Rudd and Thorson 2017). In return, to implement the above-mentioned procedure several assumptions should be accepted (e.g. life-history parameters are estimated without bias or recruitment variability from year-to-year is lower than 30%).

The second way to explore the performance of data-limited stock assessment models compares the outputs of the data-limited models with the results that a data-rich stock assessment model provides (Kokkalis *et al.* 2017). To carry out this comparison, the data-rich stock assessment model has to be previously contrasted while assumptions underlying the used models should be known enough (Harwood and Stokes 2003). This method has the advantage of allowing the assessment of a real stock, considering limitations and bias underlying input data.

Accordingly, the first step of this thesis was to determine the reliability of a data-limited stock assessment method to provide the stock indicators that often are used to advice data-rich fisheries (Chapter 2.1). For this purpose, through this thesis the European hake (data-rich species) was treated as a data-limited species, although it is a well-assessed data-rich species, assuming steady state using VIT software (Lleonart and Salat 1992, 1997). The stock assessment outputs of this variant of virtual population analysis (VPA) were compared to those derived from a variant of VPA, the eXtended Survivor Analysis (Shepherd 1999), which is routinely used to assess data-rich stocks in European waters in ICES, GFCM and STECF working groups (GFCM 2015; ICES 2015; STECF 2015, 2016).

The advice derived from the stock assessment of hake by the Scientific, Technical and Economic Committee for Fisheries (STECF) in the European Mediterranean waters is based on the trend and absolute values of harvesting (i.e. fishing mortality,  $F$ ) and stock size (e.g. recruits,  $R$  and biomass,  $B$ ) (STECF 2015). Additionally, average  $F$  of the most fished ages ( $F_{bar}$ ) is compared to a reference point ( $F_{0.1}$ ) to determine if the stock is or not under overfishing regime. In this thesis, the pseudo-cohort VPA model of VIT usually produced estimates of  $F$  lower than those estimated by XSA. This result is comparable to that found by Rätz *et al.* (2010) for the Atlantic cod when stock size was low and overexploitation was high (comparing VIT with XSA). Before interpreting this result it is worth mentioning that although the absolute values of fishing mortality estimated by XSA from 2005 to 2014 were not matched by VIT, a general agreement between the trends of annual outputs of both data-rich and data-limited methods were consistently found.

The similar annual trend on fishing mortality and stock size produced by both methods suggested that the violation of equilibrium conditions, which might have occurred during the study period, did not affect negatively the performance of the VIT software. However, a complementary interpretation of the differences between methods also suggests that even small changes in either recruitment or fishing effort (i.e. out of steady state) could promote underestimation of fishing mortality. Interestingly, better estimates of the most fished ages (1 and 2) and larger biases on estimates for less-represented ages (0 and 3) were found. These differences at age level pointed out to the single von Bertalanffy growth parameters (*vBGP*) imposed in the model to all annual length structures. In other words, the stock structure that is currently fished (GSA6, 2005-2014) and the population structure used to determine the *vBGP* (GSA7, 2006, Mellon-Duval et al. 2010), accepted in official stock assessments, may not match. This occurs because, for instance, a different truncation level of population resulted from distinct fishing efforts by area and sampling time. Thus, fishing mortality of marginal although commonly harvested ages may be unequally observed among years due to the slicing effect. Consequently, the pseudo-cohort VPA provided less biased  $F_{bar}$  when only the most fished ages were considered (ages 1 and 2).

Given that in VPA fishing mortality is used for estimating the number at age, the stock and recruits number derived from this data-limited approach were biased over similar proportions (~20%). Rätz et al. (2010) found large differences in estimates of  $B$  and  $SSB$  between pseudo-cohort VPA and XSA, probably because the larger number of age classes in cod (7 age classes in cod compared to 4 in hake) are better suited to track cohorts over time by XSA. The fishery of European hake in Spanish Mediterranean mainly harvests age classes from 0 to 3, meaning that steady-state assumptions are less forced than if more age classes are removed. Accordingly,  $F_{0.1}$  computed for most fished ages of the European hake led to lower annual biases between methods. Additionally,  $F_{bar}/F_{0.1}$  produced smaller differences between estimates of both methods, offering median relative errors lower than 30% regardless the ages computed. This result matches with a more stable annual definition of stock status when relative  $F$  is used, arguing that it removes the uncertainty on growth estimation (Kokkalis *et al.* 2017).

Larger complexity is added to the comparison of VIT and XSA when it is considered that the terminal fishing mortality ( $F_{ter}$ ) is a guessed value used as an input for starting the back calculation of fishing mortality and stock size in the VIT software. A single  $F_{ter}$  value has been used in the stock assessment of Catalan sea resources (Demestre and Lleonart 1993; Aldebert and Recasens 1996). In order to obtain the “proper” value to start the standard VPA calculations, best convergence through obtaining stable trends in the older part of the  $F$  vector was used as the main criterion. This somewhat subjective procedure, however, could involve different unquantified biases in estimates of stock indicators among years. The stock assessment of the European hake performed by VIT evidenced that  $F_{ter}$  led to an unequal confidence interval in stock size indicators among years. Years with a slightly larger contribution of smaller individuals to the stock structure tended to increase the uncertainty in stock size estimates. This means that a single  $F_{ter}$  value equally used throughout temporal analysis could lead to less realistic estimates in some years, probably related to a higher breach of steady-state assumptions rather than only true changes in the stock size.

The findings derived from a closer analysis of  $F_{ter}$  evidenced that interpreting the stock assessment outputs is more than understanding the input data limitations (i.e. potential biases) and requires an appropriate perspective about what the stock assessment results mean. Harwood and Stokes (2003) stated that all models are caricatures of reality. Consequently, the stock assessment models provide a partial, and potentially misleading, representation of the stock history. Particularly, the VPA models here used do not take into account time-varying catchability and time-varying selection patterns, implying that potential changes in fishing power along the study period were not modeled.

These caveats are even more important when data-limited models are used to interpret what might have happened at an earlier fishery stage. The historic stock assessment results suggested that the stock biomass of the European hake fell by half (mean) between the eighties and last assessed years. Bas *et al.* (1955) already warned about an unsustainably fishery pressure on the European hake in the fifties, meaning that the stock size of the European hake in GSA6 could be higher than actually recognized.

According to the findings of this thesis, the pseudo-cohort VPA may offer useful information on the trend of both fishing mortality and stock size for European hake since the eighties. Further, the comparison of the VIT model and XSA allowed to recognize the bias associated to estimates of stock indicators of European hake under steady-state assumptions. Thus, the pseudo-cohort VPA may use sparse/discontinuous data series of length frequencies, which are available at least since the early 1980s in many Mediterranean marine research centers, to help providing a more complete picture on the trend of historical evolution of fish stocks in the area.

## 3.2. Including uncertainty in stock assessment of data-limited species

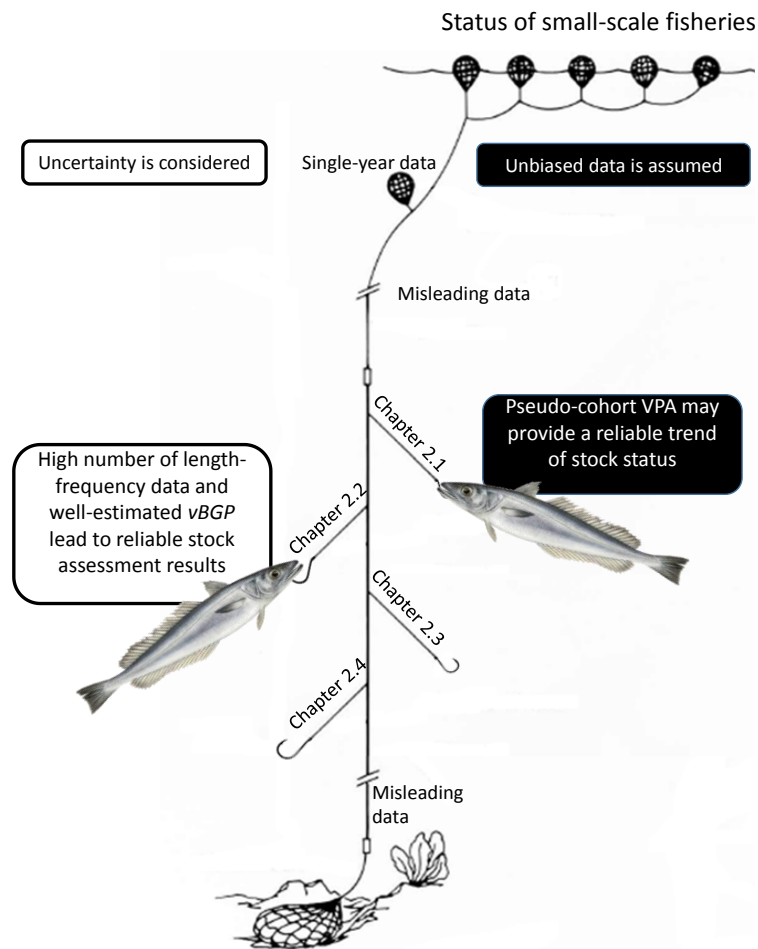


Fig. 50. Assuming that uncertainty in life-history parameters and size structure are known. Figure of “*ballestilla*” is modified from Arana and Vega (2000).

In order to use stock assessment models belonging to the VPA family, key stock information as catches and life-history parameters must be assumed as well known and unbiased (Shepherd and Pope 2002). However, data-limited species usually lack knowledge on life-history parameters (Hordyk *et al.* 2015b; Prince *et al.* 2015a). Additionally, the results of the Chapter 2.1 showed that assessment results are quite sensitive to misspecification of  $vBGP$  and  $M$ . This implies that stock assessment based on VPA methods requires a deep knowledge of both life-history parameters and harvest history. This knowledge, however, is not usually met by data-limited species. In such a situation it is essential to consider as much uncertainty as possible when setting the model, particularly when historical fishery of data-limited species is poorly known (Prince *et al.* 2015b).

Once the performance of a data-limited stock assessment method like the pseudo-cohort VPA that is fed with “unbiased” data was explored (Chapter 2.1), the next natural step was to determine the performance of data-limited stock assessment models capable to include data-input uncertainty (Chapter 2.2). Thus, the Length-based spawning potential ratio model (LB-SPR), which is also performed under steady-state assumptions, was used to test the data-limited stock assessment outputs of European hake in GSA1 (Data-rich stock). Given that the LB-SPR model has been tested through simulations (Hordyk et al. 2015a, b) and has also been used to assess data-poor Indo-Pacific resources holding small samples size (Prince *et al.* 2015b; Prince 2017), the next goal was exploring the impact of the quality of population structure on the stock assessment outputs. Additionally, this thesis took advantage of the knowledge for the life-history parameters of the European hake in Mediterranean waters to test how the uncertainty in this information affects the stock assessment results.

The concept of uncertainty in fisheries is intrinsically related to the risk in fisheries management (Francis and Shotton 1998). While the risk is the probability that a dangerous result will occur (i.e. 0 = does not occur and 1= occurs), the uncertainty considers the incomplete knowledge about a particular subject (Harwood and Stokes 2003). At least six different types of uncertainty may be differentiated in stock assessment models (Francis and Shotton 1998): 1) Process (e.g. variability in recruitment), 2) observation (e.g. related to data collection), 3) model (e.g. shape of the stock-recruitment relationship or indirect estimator of natural mortality and *M-at-age*), 4) estimation, which depends on observation and process uncertainty (e.g. number of recruits when a back-forward model of the type of VPA is used), 5) implementation (e.g. successful implementation of an annual catch limit) and 6) institutional (e.g. problems associated to relationship between stakeholders). Higher complexity may be considered in a fishery management cycle when the unexpected behavior of fishers is also incorporated as a source of uncertainty (Fulton *et al.* 2011).

Chapter 2.2 was mainly focused on the uncertainty related to observation and estimation, although the process uncertainty was, by extension, considered. The other three sources of uncertainty (model, implementation and institutional) were out of the scope of this chapter. The estimation uncertainty was incorporated in the stock

assessment of the European hake by considering four different growth models based on the von Bertalanffy growth equation (this means that it was assumed that the von Bertalanffy growth curve properly describes the growth of the European hake). The estimation uncertainty was additionally included from triangular distribution of the values of natural mortality and maturity at size that were used by the official stock assessment of this species (STECF 2015). The assumption on the uncertainty distribution is relevant since the biological traits of the European hake are assumed as well-known. For this reason, this stock was assessed as data-limited instead of data-poor species, according to the stock assessment categories proposed in this thesis.

The slow-growth hypothesis ( $k$  values of approximately  $0.1 \text{ yr}^{-1}$ ) and fast-growth hypothesis ( $k$  values of approximately  $0.2 \text{ yr}^{-1}$ ) used in this study (Mellon-Duval *et al.* 2010), did not exceed 10% of the asymptotic length estimated for the Murcia region ( $L_{\infty} = 105.87 \text{ cm}$ ). Thus, the probabilities of underestimating or overestimating  $F/M$  or affecting the estimate of SPR due to inaccurate or imprecise estimates of  $L_{\infty}$  were low (Hordyk *et al.*, 2015c). On the other hand, the indirect estimator of natural mortality (PRODBIOM) used by the official stock assessments of the European hake (Abella *et al.* 1997; STECF 2015) strongly depended on the  $k$  value ( $r^2 = 0.87$ ). This result implies that the concept of the fast- or slow-growth hypothesis virtually disappears when  $M/k$  instead of  $M$  and  $k$  separately is used. Thus, two growth hypotheses produced fairly similar  $M/k$  values. Variation in SPR estimates did not depend on the life-history parameters assumed but on the length structure used to describe the exploited population.

The observation uncertainty associated to the length structure of a population harvested by fisheries may be addressed in two main ways. Resampling the available data to produce different sets of length structures that posteriorly will be tested in the stock assessment model or comparing the performance of the stock assessment model under different length-frequency sizes (e.g. comparing different years). Chapter 2.2 used the second choice for several reasons. First, the length structure of the data-limited and data-poor stocks usually did not represent the full population in the fishing area because gear selectivity promotes that some ages are more fished (Huse *et al.* 1999; Garcia and Ramírez 2016). Second, many times the oldest individuals are poorly represented

because the population is largely truncated (Hixon *et al.* 2013; Saari *et al.* 2014). Third, several species caught by small-scale fisheries exhibit low representation in landings (Prince *et al.* 2015b). If the length structure is subjected to resampling, all the above-mentioned problems will be emphasized obtaining final length structure probably far away from the population wanted to be assessed.

A better performance of the LB-SPR model was found for data-poor species when the length structure that fed the stock assessment model was higher than 1000 individuals (Hordyk *et al.*, 2015c; Prince, 2017). However, these authors did not test well-assessed species holding larger sampling size. Findings of this thesis indicated that even though our sample sizes were larger than 1000, the SPR estimates from the four life-history parameter sets varied among years (from 1% to 6% approx.). In the framework of fisheries management using a harvest control rule, even small changes to the current SPR could mean additional time to return to sustainable levels because more than one recovery trajectory may occur (Goodyear, 1993; Hordyk *et al.*, 2015a). Seen from the socio-economic side of the small-scale fisheries, a higher reduction of fishing effort or larger periods of closure may mean larger implementation uncertainty or in an extreme case the failure of the enforcement of the management measures.

The years when the four life-history sets produced virtually the same SPR values were derived from large samples (> 2000 individuals) and corresponded to most of the best fits to the annual length distributions. These results suggest that if the length structure properly represents the population  $M/k$  derived from either fast or slow growth lead to similar LB-SPR estimates. Accordingly, the stock assessment of data-limited species should prioritize collecting as large as possible number of length samples by month for medium or long-lived species or even daily for short-lived species. Once that the impact of estimation and observation uncertainties were established to a data-moderate species as the European hake, the process uncertainty was explored. This type of uncertainty is rarely known in data-limited, and much less, in data-poor species. The recruitment variability ( $\sigma_R$ ) of the European hake in the West Mediterranean (GSA: 01, 05, 06 and 07, see STECF, 2015) from 2003 to 2014 was estimated between 0.33 and 0.41. Hordyk *et al.* (2015c) found that the LB-SPR method performed relative well when  $\sigma_R$  is lower than 0.6, producing estimates within 30% of the true value. This result

matched the lower variation of  $F/M$  and the comparable values of the SPR estimates when the size compositions were grouped into three- or four-year periods. Thus, moderate changes in recruitment, which probably led to transitory population size structures, did not prevent the reliable SPR estimates for hake.

The European hake individuals removed by the bottom trawler fleet in the Murcia region (GSA1) were of sizes much less than the asymptotic length. In addition, length of selectivity ( $L_s$ )  $L_{50}$  and  $L_{95}$  estimated by LB-SPR correspond to 37% and 47% of the length-at-maturity ( $L_{50} = 30$  cm), respectively. Under this fishing regime, promoting catches of immature individuals and absence of oldest (or largest) individuals, the SPR remains at or below 8%. Probably, the stock in the Western Mediterranean Sea, although continuously decreasing, has not collapsed yet because large individuals may be unavailable to the fishing gear and protected in “spawning refugia” (Caddy 2015). This would imply that both conventional (STECF, 2015) and data-limited (e.g., this study) stock assessment methods would overestimate fishing mortality and underestimate spawning biomass.

If the oldest individuals are not available, the length structure of the population that is used to determine the stock status is biased (e.g. toward gear schedule). In the particular case of the European hake,  $M/k$  was found higher than 2.3, suggesting that the unfished stocks is dominated by young classes (Hordyk et al., 2015b). These  $M/k$  values may be compatible with a “natural” stock that is mainly represented by young individuals. Stock assessments for this kind of population, with low representation of adult classes, would be less sensitive to the possible bias introduced by ignoring dome-shaped selectivity (Hordyk et al., 2015c). However, the possibility of dome-shaped selectivity cannot be excluded (Mahévas et al., 2011). Specifically, diel vertical migrations of larger individuals and/or their preferential distribution in non-trawled areas may result in dome-shaped selectivity because Spanish Mediterranean trawls are permitted to operate during daytime hours only.

## 3.3. The role of participatory monitoring in small-scale fisheries

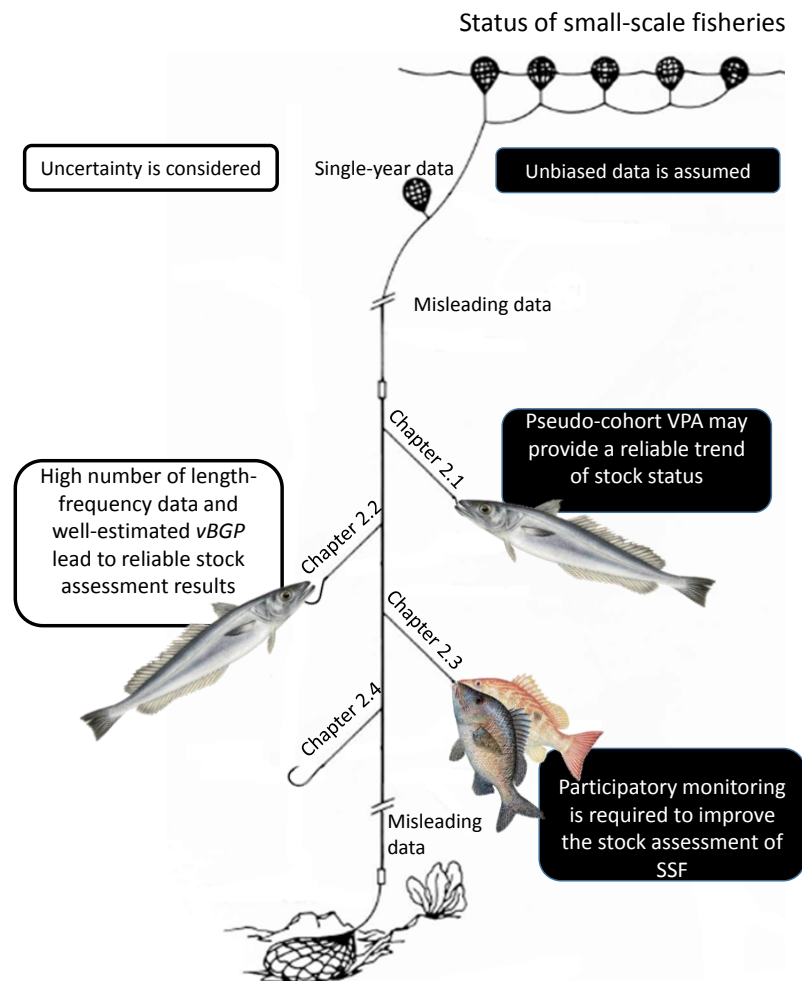


Fig. 51. Participatory monitoring helps to reduce biases on input data of the fishery targets. Figure of "*ballestilla*" is modified from Arana and Vega (2000).

The chapters 2.1 and 2.2 addressed some of the most important problems associated to the stock assessment under the steady-state assumptions, which are suitable to determine the status of small-scale fisheries lacking catch history. However, the intrinsic characteristics of the European hake as a data-moderate resource (e.g. large number of sample size on length structure, well-known life-history parameters, most of the catches landed in known fishery ports and availability of some information on recruitment variability) hinders the understanding of other challenges faced for assessing data-poor small-scale fisheries. Chapter 2.3 of this thesis leaves the "lifejacket" until now used to assess a data-moderate species under a data-limited framework (The bottom trawl fishery of the European hake in the Spanish Mediterranean). After the first

two chapters, this thesis focused on assessing fishery resources holding only some pieces of data, probably biased, because they may only recover a small part of the information of the “original” stock (data-poor fisheries). The latter condition is the most common case among small-scale fisheries (Salas *et al.* 2007; Teh and Pauly 2018). One of the objectives of this thesis was to examine whether reliable stock assessments can be performed at all in these less than ideal conditions.

A formal stock assessment of any resource relies on good estimates of life-history parameters (when required) and catch data. However, small-scale fisheries usually land in rural areas that many times lack logistics and trained teams to record catches and full biological information (Salas *et al.* 2007). In order to understand and address the problems associated to misleading landings, the two most important fish resources harvested by the Wayuu fishery in the Colombian Caribbean were assessed (as case study). The reconstructed landings using information derived from participatory monitoring tended to decrease (2006-2014), but this behavior was not detected by the official monitoring. Additionally, the most-used gears (gillnets, encircling gillnets and longline) showed different level of underestimation each year. Biomass removed by the encircling gillnet fleet was landed in urban areas until 2010. From that point, catches were landed in both urban and rural areas. Consequently, the official monitoring recorded only a portion of the entire harvest for each year. As a result, an unpredictable bias in the estimation and trend of stock size may occur (Omori *et al.* 2016; Rudd and Branch 2016).

The above-mentioned findings demand thinking on how a stock harvested by small-scale fisheries may be properly assessed if the fishing effort and stock size have changed in an unknown way. Particularly, at least three different regimes of fishing effort on the lane snapper have been identified since 1982 in the Colombian Caribbean (García, 2010; Manjarrés, 2004). A part of the two last regimes were recorded during the study period (2006-2009: “longline” regime and 2010-2014: “encircling gillnet” regime). Nevertheless, the regime shift did not occur at the same time in all fishing communities. Accordingly, it was required to use several raising factors to separate the effects of port, fleet and period (i.e., month and year) on the reconstructed landings. This procedure is substantially distinct to that carried out to reconstruct the marine

fisheries landings in Colombia and other developing countries, which has used a single raising factor for all time periods (Lindop et al., 2015; Wielgus et al., 2010; Zeller and Pauly, 2007). Consequently, the differences in harvesting power among fleets and the effects of different fishing effort regimes on the reported landings is ignored, leading to an misleading estimate of fishing pressure and population status.

Although the stock assessments of the lane snapper and white grunt in the Colombian Caribbean did not consider the observational uncertainty associated with landing values, through including the participatory monitoring, the bias in catch history was considerably reduced. Thus, in a broad sense, the uncertainty on the trend of landings decreased to levels such that stock assessment becomes really informative on the stock status. The von Bertalanffy growth parameters of data-poor species obtained from fishery-dependent data in this study are probably biased. This occurred because continued overfishing usually leads to age truncation (Kenchington, 2013; Saari et al., 2014). Furthermore, the fishery-dependent length structure used to estimate the growth parameters underestimated the smallest and largest fish. Indirect estimators of  $M$  propagate the bias from the  $vBGP$  in calculating natural mortality (Kenchington, 2013; Then et al., 2015). Given that the population status depends on  $M$  value that is used in the stock assessment model, even contradictory information about stock status may be derived from a same fishing effort when one or other value of  $M$  is used.

While the importance of the  $M$  value in stock assessment is out of doubt, the pseudo-cohort VPA works under the “constant parameter system” (Sparre *et al.* 1989), meaning that among others  $M$  must be assumed as known and constant. However, few authors have conducted studies to compare the performance of indirect estimators of  $M$  and reasonable estimates of  $M$  from wild populations (Gislason et al., 2010; Kenchington, 2013; Newman, 2002; Then et al., 2015). Most argue that estimators based on maximum age ( $T_{max}$ ), mainly Hoenig’s estimator, should be selected instead of life-history correlation estimators (Kenchington, 2013; Then et al., 2015). Particularly in the case of snapper, exhibiting fast initial growth and later move toward slow growth, Pauly’s estimator proved to perform badly (Kenchington, 2013).

When different “families” of indirect estimators of  $M$  (e.g. maximum-age based or  $vBGP$  based) were used to estimate  $M$  of lane snapper and white grunt, it was noted

that both stocks may exhibit an overexploited and sustainably exploited status at the same time. Healthy stocks seem unlikely to occur in the Colombian Caribbean because fishing mortality of small-sized fish between early and later years tended to increase, while landings and CPUE were strongly reduced. The annual CPUE of the encircling gillnet vessels, including all caught species, was ten and eleven times higher than those of the bottom gillnet and long-line vessels, respectively. During the study period, the annual-average catch per vessel fell by 20.40% (encircling gillnets), 48.06% (bottom gillnets) and 75.27% (longline vessels). Supported by the latter information is more feasible that overexploitation of both species is occurring. However, definition of fishing mortality under overexploited conditions should be carefully used because overestimation of current fishery pressure would also lead to a large underestimation of fishing mortality in the long term (Clark, 1999).

Contrary to what occurred when  $M/k$  instead of  $M$  and  $k$  individually are used to set the stock assessment model (e.g. using LB-SPR, chapter 2.2), the stock assessment performed by pseudo-cohort VPA is conditioned by either the fast-growing or slow-growing hypothesis (Mellon et al., 2010). An increase in  $k$  means a reduction in longevity, affecting indirect estimates of natural mortality (ICES, 2014). Variation in the  $vBGP$  also led to large changes in the levels of  $F/F_{msy}$  and  $SSB/SSB_{max}$  for both species. Lane snapper and the white grunt distributed in the Caribbean Sea and East Atlantic Ocean exhibit fast- and slow-growing (see Methods of chapter 2.3 to find all references). Nevertheless, estimates of these growth parameters have been based on different methods and populations with diverse exploitation levels, meaning large uncertainty in those estimates. Such a level of uncertainty in the stock assessment of these data-poor species was addressed by testing both growth hypotheses. The results indicated that a higher  $k$  for lane snapper produced greater changes in the overexploitation level throughout the study period. Greater changes between years would be desirable (although its value cannot be accurately estimated), given that the responses of this population to unexploited or slightly exploited pressures are unknown.

The estimates of virgin biomass and stock size of both species were affected by 1) the steady-state assumptions underlying the stock assessment model, 2) the need of assuming that changes in gear selectivity did not occur during the study period and 3)

strong year-class effect that occurred in years with outstanding reconstructed landings (e.g. landings of white grunt in 2010). These conditions, along with the uncertainty in the estimation of the  $vBGP$ , probably contributed to masking the total effect of a landings decrease on the trend of stock size. In other words, although stock assessment has been performed under reasonable assumptions on input data, the outputs uncertainty is difficult to quantify. Under these conditions, stock assessment of data-poor stocks should prioritize determining the uncertainty associated to estimates of  $vBGP$ . Accordingly, monitoring programs that consider the life-history of the target species are crucial to reduce uncertainty in estimates of  $vBGP$  by length-based methods and validate them through otolith readings when possible.

At this point, to perform stock assessment of data-poor fisheries, important decisions must be taken. In order to obtain or improve the key stock parameters and subsequent status of data-poor stocks, Punt et al. (2011) outlined among-stock penalties when these stocks were jointly caught with data-rich stocks. Nevertheless, most of the stocks of small-scale fisheries in developing countries are data poor, meaning that the abovementioned approach cannot be implemented. From a derivation of the concept of “borrowing” information of data-rich stocks to assess data-poor stocks, other authors have used the best available information on life-history parameters to set the stock assessment models under data limitations (Kokkalis et al., 2015; Prince et al., 2015). Thus, the concept of “Robin Hood” approach was coined at the assessment of data-poor fisheries when some kind of information is “borrowed” from data-rich stocks (Scandol et al. 2009; Smith et al. 2009). As was extensively explained in the introduction subsection 1.6 (“Robin Hood” approach), the use of life-history parameters of related fished stocks to set a particular stock assessment model is not a straightforward practice (deeper explored in chapter 2.4).

The available evidences on fishery of lane snapper and white grunt suggest that fishing mortality was higher than the set of natural mortality values used to set the stock assessment model of both species. The biases and uncertainty underlying estimates of the life-history parameters used in stock assessment of these species allowed understanding the scope of the stock assessment outputs. Therefore, the stock status, in these particular cases of overexploitation, may be provided using a range of values

(e.g.  $F/F_{msy}$ :  $1 < \text{overfishing} < 3.5$  for lane snapper or  $SSB/SSB_{max}$  overfished 5–18% for white grunt).

The above-mentioned results provide valuable information about what may and may not be advised about the status of data-poor fisheries. An accurate level of overfishing as well as the spawning stock biomass that remains under the current fishing pressure are difficult to determine. This is all the more so pseudo-cohort VPA assumes that input model data is known but it is fairly sensitive to variation on life-history parameters and catch data. Therefore, advising the small-scale fisheries from the stock-indicators trend more than their absolute values should be preferred. Additionally, the knowledge on the biases in input data did not necessarily lead to reduce the uncertainty in the stock assessment outputs. While the knowledge of biases in input data help to determine the scope of the stock assessment results, uncertainty reduction (for some type of them) only is possible when input data is consciously improved.

Once the model was set with the best available information, large differences in stock status were found when stock assessment models fed with only official landings and official plus participatory landings were compared. The official landings of lane snapper led a higher overexploitation status than when the full landings information was used. A large underestimation of landings in rural areas for longline and gillnet fleets by the official monitoring program produced small variations in landings during the study period. Thus, a false view of the selectivity pattern of the lane snapper fishery was provided. These results indicate that the advice derived from the stock assessment that was fed with only official data increases the collapse risk of the lane snapper population. This will occur despite the fact that a higher overexploitation status has been indicated. A stock assessment that demonstrates sustained overexploitation while low changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$  among years occurs would mean that the population timeframe observed (2006–2014) was quite stable. This can lead to the false assumption that the lane snapper population has been capable to absorb the overfishing effects.

From adding participatory monitoring, it is highlighted that in nine years the  $SSB/SSB_{max}$  could have been reduced by half, while  $F/F_{msy}$  could have been doubled. Although the stock assessment performed using only official data produced higher and sustained levels of overfishing (meaning overestimation of  $F$  during the study period), it

does not necessarily help to decrease the risk of collapsing. Conversely, full landing information allowed to observe a faster stock declining than estimated by using only official information. The stock assessment of the white grunt provides an *a priori* more complex stock status interpretation given that the two sources of landings did not lead to distinct outputs of  $F/F_{msy}$  and  $SSB/SSB_{max}$ . This is because the encircling gillnet fleet was the dominant gear and presented a similar selectivity pattern that the second most used gear (gillnets). Therefore, the selectivity pattern of the white grunt fishery was not significantly affected if landing points were altered. This implies that determine the maximum sustainable yield (e.g.  $F/F_{msy} = 1$ ) requires an entire knowledge of the total landings by fleet. This, however, is only feasible if landings recorded by the participatory monitoring program are considered in the stock assessment and further management simulations.

As expected, the uncertainty in the life-history parameters and bias in the landings information of small-scale fisheries may be easily propagated into the management simulations. Using the best available information and assuming two opposite recruitment models, moderate uncertainty was found in the simulation outputs (lane snapper, 9%-33%; white grunt, 29%-40%). According to stock assessment findings, the lane snapper and white grunt stocks are largely overexploited, meaning that recruitment may be compromised. Considering the management measures included in simulations, the Wayuu fishery of lane snapper and white grunt may recover after eight years if the encircling gillnets are entirely eliminated. The weak socio-economic status of Wayuu people probably makes impractical the enforcement of more restrictive management measures (Ramírez 2015). The management simulations of small-scale fisheries are a useful tool for advising, for instance, what fleet should be reduced or eliminated for promoting stock recovery. Nevertheless, what reduction level or during how long should it be enforced are questions hard to answer, especially when implementation uncertainty usually is high and the behavior of fishers is not entirely understood (Fulton *et al.* 2011).

3.4. Assessing data-poor small-scale fisheries

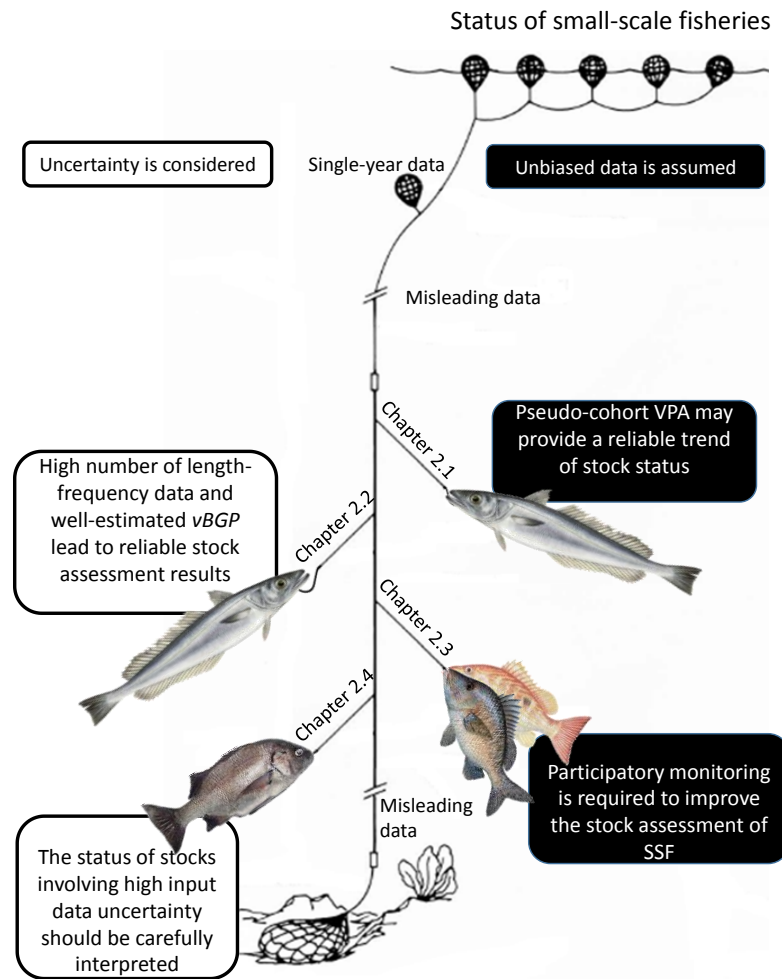


Fig. 52. Data-poor species involve high level of uncertainty that hinder an accurate definition of stock status. Figure of “*ballestilla*” is modified from Arana and Vega (2000).

Only some pieces of the full information required to manage complex SSF fisheries are available to scientists, managers and even to fishers themselves. When a stock declines, fishers explain such stock depletion by the intervention of external factors (e.g. atypical weather, pollution or excessive tourism) before accepting that they could play an outstanding role in that declining (García 2010; Ramírez and Puentes 2015). Fishers blame other fishers who fish with different gears and techniques than those used by the questioning group as part of external problems (Fig. 53). In other words, the own responsibility is the last accepted explanation to a stock declining.

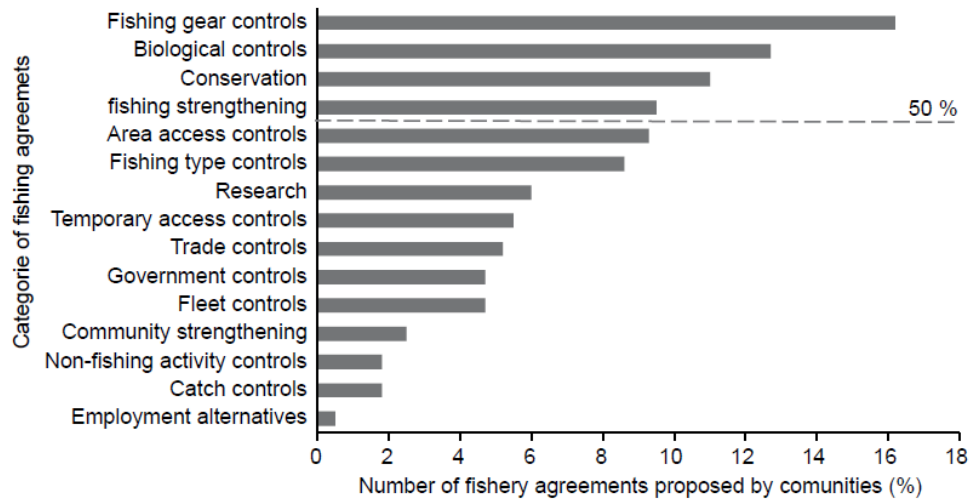


Fig. 53. Percentage of total number of fishing agreements proposed by Colombian fishermen ( $n=36$ ) in order to solve the problems around fishery activities from a community perspective. Note that catch controls (biomass) are not prioritized by fishers while controls on “selectivity” (gears), size of catch (biological) were importantly recommended. Modified from Ramírez and Puentes (2015).

The absence of a critical view of fishers about the reasons of stock declining is translated into lack of uniform criteria to record and interpret the catch history (by either landings monitoring or traditional knowledge). The fishery managers usually have a centralized point of view, while participatory management (or any of the forms of co-management) is still timidly enforced (Ramírez and Puentes 2015). Most of the catches landed in rural areas are not recorded by the official monitoring (Salas *et al.* 2007). Additionally, the official management is based on biased information that do not cover the full condition of the stocks that are managed (an example in Fig. 54). The scientists use formal methods to collect “unbiased” data, but these data is referred to only a portion of the full fishery history. The fishery science uses information as those described above to model the trend and status of the stock. Not surprisingly, the stock assessment outputs of small-scale fisheries involve large uncertainties that should be properly interpreted (Dowling *et al.*, 2019).

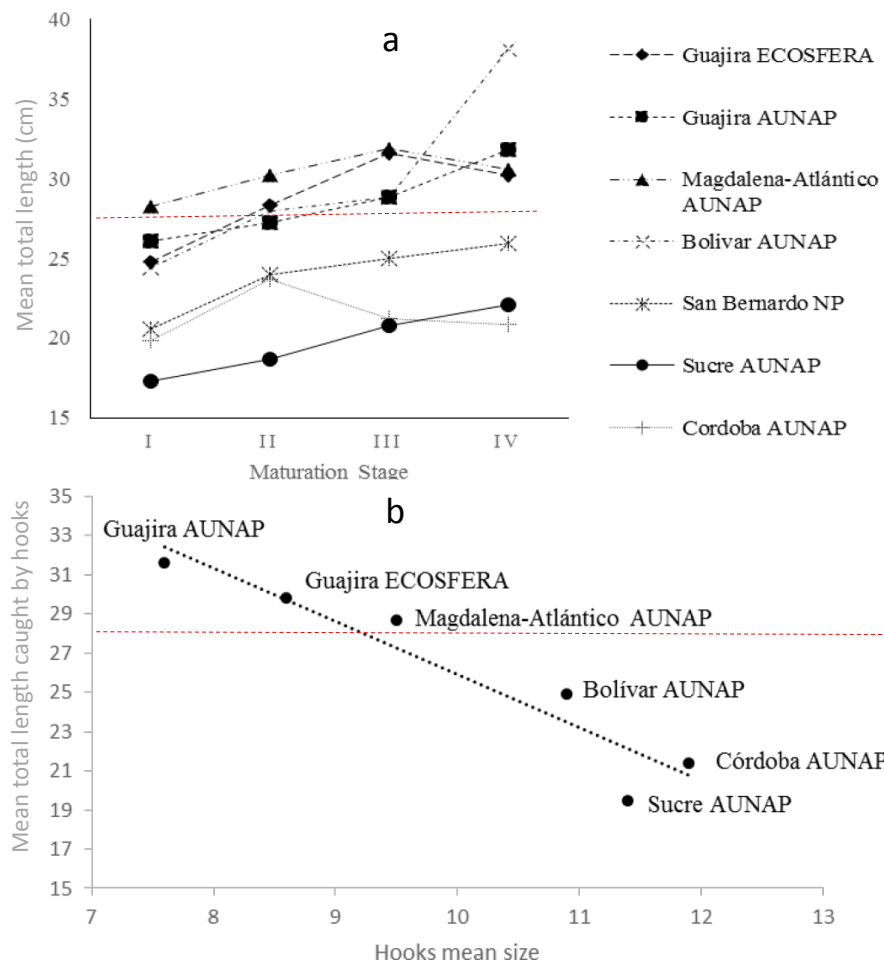


Fig. 54. Perceived size of maturity of lane snapper in the Colombian Caribbean. a) Mean size per maturity stage by the region where individuals were fished. b) The relation between length at first maturity ( $L_m$ ) and mean total length (TL). Modified from Garcia and Ramírez (2016). The official size of maturity used by the government to manage the whole stock of lane snapper in the Colombian Caribbean (red dotted line) (Aunap-Universidad del Magdalena 2013). Note that the same maturity stage could be categorized as mature and immature (by the official size of maturity) depending on the place where the samples are coming from. Additionally, the perceived size of maturity relied on the fishing gear (method of sampling) used to collect samples.

Chapter 2.1 demonstrated that stock assessment performed by pseudo-cohort VPA using sparse data under steady-state assumptions may provide reasonable trend on the size and status of data-limited stocks. However, the model input data was assumed unbiased, very unlikely in small-scale fisheries by the reasons explained along the chapters 1, 2 and 3 of this thesis. The chapter 2.2 accounted with uncertainty derived from life-history parameters and sample size for a data-limited species relatively well known. However, the sample size used to carry out the research of this chapter is not available for most of the resources harvested by small-scale fisheries. The chapter 2.3 is used to highlight the importance of participatory monitoring to improve the information on life-history parameters (used to set the stock assessment models) and catch history

of data-poor resources. At the same time, this chapter demonstrated that the stock assessment model performed under steady-state assumptions (pseudo-cohort VPA) is quite sensitive to misspecification of life-history parameters. Given that the common denominator of the three first chapters (2.1, 2.2 and 2.3) was the large impact of life-history parameters, and particularly, the growth information (here assumed well described by the von Bertalanffy function) chapter 2.4 was dedicated to deeply explore the uncertainty in both the selection of von Bertalanffy growth parameters and the assumptions related to their distributions.

The Peruvian grunt is the second most important beach fishery resource after the fine flounder in Peru (INEI 2012). Surprisingly, it is assumed a single growth curve along the Peruvian coast (latitudinal gradient encompassing from 4° to 18° south) (IMARPE, 2018), while an outstanding effect of water temperature on the growth of this species was already evidenced (Dionicio-acedo et al., 2018). Additionally, information on growth does not exist in global fish bases (e.g. FishBase, Froese and Pauly 2018). The case of the Peruvian grunt is not an isolated case of lack of information of fishery resources harvested by small-scale fisheries. Probably, it is still more common case than desirable one hundred years after Hjort set the foundations for the demography of fish for assessing fished populations (Hjort and Lea 1914). At this respect Pauly and Zeller (2017) noted that the “assessed world” fisheries really belong to well-known stock in developed countries while most of the small stocks in developing countries have never been assessed. Thus, the Peruvian grunt provides a good opportunity for understanding what may be advised when there is a limited or poor information on life-history parameters and no data on catch history, catch at age data, fishery-independent data nor reliable CPUEs.

Through the monitoring of the beach fishery of the Peruvian grunt was identified that estimates of CPUE may be largely biased because of many different fishing gears are used (holding several gear schedules), fishing activity is quite variable along day, at night and among seasons, and oceanographic conditions at a microscale continuously change. Given that the Peruvian grunt is associated to breaking zones (Angel and Ojeda 2001; Perez-Matus *et al.* 2007), the fishery performed onboard small boats avoid to land in this kind of beaches. Consequently, the official monitoring that is focused in fishing

ports do not properly record the Peruvian grunt fishery (INEI 2012). Thus, catch history provided by official reports is doubted (Guevara-Carrasco and Bertrand 2017). It has been recognized a wide fed plasticity on this species, exhibiting diets from totally herbivore until totally carnivore depending on the prey availability (Pérez-Matus *et al.* 2012; Malay *et al.* 2015). Stock production may be controlled by the habitat (Brown *et al.* 2018), because among others, growth depend on quality of the available feed. Accordingly, the participatory monitoring was used to determine the length structure of the fished population, estimate the von Bertalanffy growth parameters (including uncertainty) using local data and stablish the status of the stocks based on a length-data model (LB-SPR).

The beach fishery of the Peruvian grunt is focused on mature individuals that are abundantly harvested by different type of gillnets. Given that there are no records of the largest size of Peruvian grunt at the early stage of this fishery, there is lack of knowledge of maximum observed sizes of this species. Despite that the stock distribution and the fishing effort on this species overlap, some of the largest individuals may remain unfished given the gear's selectivity. These potential biases in the data length structure used to estimate the  $vBGP$  by length-based methods may affect the  $M$  estimates using indirect methods based on  $vBGP$ . This thesis results indicated that the definition of the LB-SPR of the Peruvian grunt depends on how narrow the life-history parameters range is.

The Peruvian grunt, as occur in many data-poor species, does not count with direct ageing analysis. Additionally, largest individuals may be already removed, meaning that the maximum observed age currently observed and that reached by the unfished population may be distinct. Therefore, several  $M$  estimates based on maximum age but also other parameters (e.g.  $vBGP$  and temperature) were used to set the  $M$  uncertainty in the stock assessment model. Uncertainty on  $L_{\infty}$  and  $M/k$  ratio were explicitly accounted into the biological model in an operating model framework as outlined by Harwood and Stokes (2003). An acceptable representation of reality could emerge from a convergence of biases that the whole life-history parameters distribution underlies. However, several sources of life-history parameters data produced diverse stock statuses (both above and below  $SPR = 0.4$ ). This means that depending on what

life-history parameters were used, the stock could be categorized as overexploited or sustainably exploited.

Contrary to the case of the Wayuu fishery in the Colombian Caribbean where life-history parameters, information on CPUE and some catch history was available to lane snapper and white grunt stocks, there is not complementary information (e.g. CPUE) that may be used to settle the more plausible stock status of the Peruvian grunt. Under these circumstances, it seems reasonable to assume that all  $L_{\infty}$ - $M/k$  combinations have equal contribution in the LB-SPR model. Thus, it is not required to define the “right” uncertainty distribution of the life-history parameters. Under such assumption, the SPR distribution was defined as overexploited, showing two main peaks in 0.10 and 0.25. The lowest SPR peak was produced by an unfished population with a median  $M/k=1.5$ . This ratio fits the concept of the Beverton and Holt life-history invariant BH-LHI (Hordyk *et al.* 2015a). The second SPR peak could be produced by an unfished population with lower proportion of large individuals ( $M/k=2.2$ ). This latter  $M/k$  also means that the absence of largest individuals penalize less the SPR estimate because large individuals were not entirely removed by fishery.

Regardless of what  $M/k$  was assumed as more feasible to represent the unfished stock of the Peruvian grunt, the proportional participation of adult classes was low ( $M/k>1.5$ ). However, there are not enough arguments to attribute one or another SPR peak to the current stock status. As the stock looks overexploited, the lower peak (e.g.,  $SPR_{2014} < SPR_{0.2}$ ) may be considered as a precautionary status, while higher peak (e.g.,  $SPR_{0.2} < SPR_{2014} < SPR_{0.4}$ ) will allow the enforcement of less-drastic management measures for recovering the SPR. Accordingly, the ratio of  $M$  to  $k$  that is used to set the LB-SPR model is responsible to determine the shape of the unfished stock structure that will be used to contrast the current stock structure of the stock exposed to fishery. However, it is the asymptotic length the parameter that helps to determine how exploited is the stock.

The LB-SPR model assumes asymptotic selectivity (Prince *et al.* 2015b). Then, when the portion of adults that is expected according to  $M/k$  value is far from the  $L_{\infty}$  used to set the LB-SPR model, the stock is more overexploited than if adult individuals being close to asymptotic length are part of length structure of the fished population.

Most of fishing gears used by these kind of fisheries (e.g., hooks and gillnets) usually exhibit dome-shape selectivity (Huse *et al.* 1999; Kuparinen *et al.* 2009). Potential biases in the SPR outputs by either a misleading representation of the stock length structure or dome-shaped selectivity were considered low. However, it is difficult to make sure that all adult-size classes could be equally fished, meaning that SPR output could be slightly underestimated.

It is common to use the triangular distribution to set the distribution uncertainty of parameters required by the stock assessment models (Hordyk *et al.* 2015a; Scott *et al.* 2016). The triangular distribution uses a guessed value to explain the highest probability of occurring (e.g. mean, mode or median) and also determine the values where that parameter becomes zero. In the chapter 2.2 of this thesis the triangular distribution was used to set the uncertainty distribution of the life-history parameters of the European hake. In that particular case, it was a reasonable decision because there was information enough to set the lowest, middle and highest values of each parameter, although the real uncertainty distribution was not known. Conversely, no reliable information was available on neither life-history parameters nor historical fishing effort of the Peruvian grunt. That implies that through using the triangular distribution, the natural uncertainty on the distribution of the life-history parameters is forced to produce a single peak.

Although the single peak found by the triangular distribution indicated overexploitation, the uncertainty distribution depended on unavailable information. It was confirmed when the lowest and highest values of that distribution were constrained by eliminating the life-history values that on a subject way were considered less probably of occur (e.g., too small  $L_{\infty}$ ). Both the uncertainty distribution that assumed equal contribution of all life-history parameters estimated and that based on triangular distribution did not entirely represent the life history and available information of the Peruvian grunt. The Monte Carlo simulation offered an SPR distribution supported by more biological sense because the uncertainty distribution depended on what was the most represented range of the life-history parameter values found regarding the full life-history parameters. In other words, those values, for instance, of  $M/k$  that were most common had a higher probability of contributing to the uncertainty distribution than

those poorly represented. Thus, the parameters distribution produced by Monte Carlo simulation emphasized the overexploited condition of the Peruvian grunt.

Given that the Monte Carlo simulation led to three main peaks below the reference point ( $SPR_{40\%}$ ), it was required to investigate the source of such results. Uncertainty on  $M$  estimates produced larger impact than uncertainty in estimates of  $vBGP$ . Eight of the nine  $vBGP$  sets led to overexploitation statuses ( $SPR < 0.4$ ). Only the “Robin hood” approach that used the Burrigo grunt data (Cruz-Romero *et al.* 1993) revealed a scattered SPR distribution around 0.5 (tending to sustainable exploitation). These higher SPR outputs occurred because  $L_{\infty}$  was 20% lower than the mean of the other eight asymptotic length estimates and the largest individuals caught by the beach fishery were 14% larger than the asymptotic length of Burrigo grunt. This means that the Peruvian grunt would be characterized by a stock that reaches the asymptotic length faster than the other eight cases and consequently, individuals that reach the maximum estimated size were relatively abundant. However, this results also indicated that the borrowed life-history parameters of a taxonomically-related species as Burrigo grunt (same genus), did not properly represent the growth of the Peruvian grunt.

The analysis of the life-history parameters uncertainty indicated that  $M$  values contributed more than  $vBGP$  to a scattered SPR distribution. None of the nine LB-SPR models (according to selected  $L_{\infty}$  and set of  $M/k$  used) produced an SPR distribution around a defined value. Conversely, the SPR distribution usually ranged values that belong to two different stock status at the same time (e.g.  $0.2 < SPR < 0.2$ ). This apparently contradictory SPR occurred because the natural mortalities used to set the  $M/k$  ratios led to different kind of populations according to contribution of adults of the unfished length structure of the stock. When the routine of *FishLife* package updated the life-history parameters estimates with local data (length based), the SPR distribution tended to be more grouped ( $0.05 < SPR < 0.25$ ). These results probably confirm one of the most important findings of this thesis because data-poor resources harvested by small-scale fishery that 1) lack knowledge of life-history parameters, 2) present an unknown uncertainty distribution of the life-history parameters and 3) do not hold reliable catch history, probably cannot be accurate assessed.

The SPR outputs derived from the three *FishLife* routines performed suggested a large variability in the life-history parameters estimates. It is hypothesized that whether taxonomic-related species are not well represented in *FishBase* or their life-history parameters largely differ between taxa, the estimates for a given not studied species (as Peruvian grunt) may provide biased estimates from global data. Finally, a higher contribution of larger individuals to the stock structure, as shown in 2014, led to larger SPR dispersion. These results suggested that an inaccurate parameterization of  $M/k$  ratio penalizes better-shaped stocks. On the other hand, the Peruvian grunt assessment indicates that local length-based data is need to improve the accuracy of life-history parameters estimates when data from worldwide fish database (Froese and Pauly 2018) are used.

## 3.5. Key features from stock assessment of data-poor small-scale fisheries

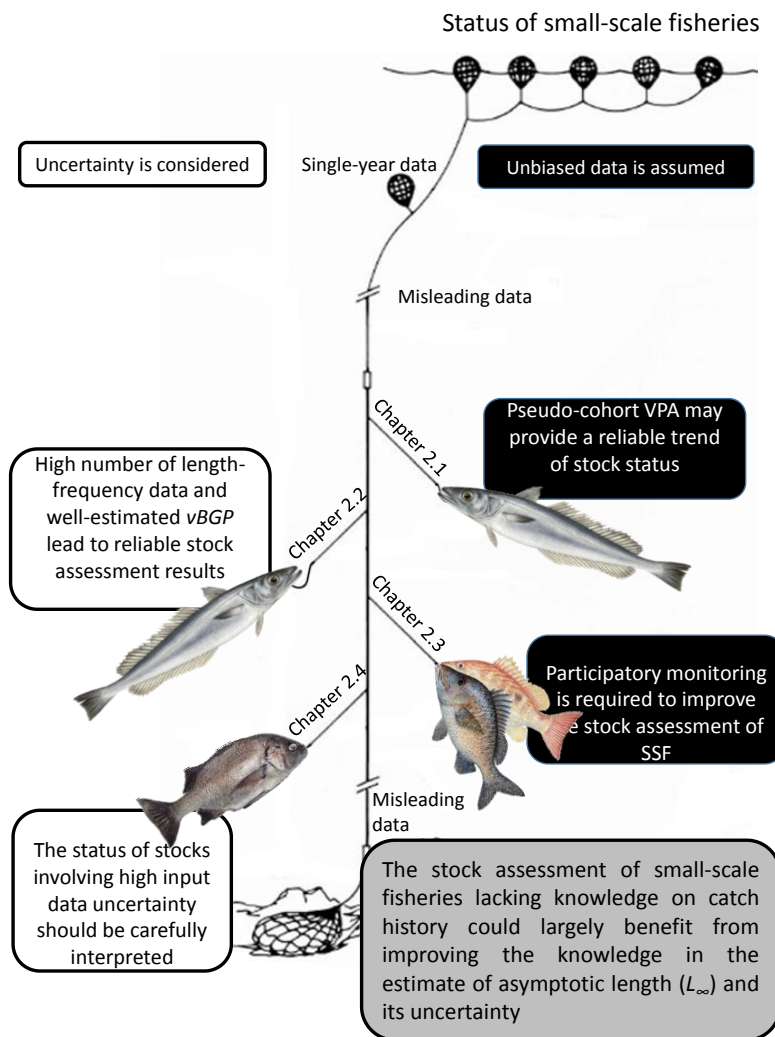


Fig. 55. Improving the life-history parameters of data-poor stocks is prerequisite to provide a proper advice of the status of small-scale fisheries. Figure of "ballestilla" is modified from Arana and Vega (2000).

Before formally presenting the conclusions obtained through developing the aims of this thesis, it is appropriate to summarize and shortly explain, from my point of view, what the key finding of this work is. The stock assessment of small-scale fisheries lacking knowledge on catch history could largely benefit from improving the knowledge in the estimate of asymptotic length ( $L_{\infty}$ ) and its uncertainty. The stock assessment models capable to deal with annual data in isolation (assuming steady state) are quite sensitive to misspecification of life-history parameters as was observed in this thesis and also by other authors (Prince *et al.* 2015b; Kokkalis *et al.* 2017; Taylor and Mildemberger 2017). Appropriated estimates of  $L_{\infty}$  helps to determine how truncated the stock

structure of a fished population may be. However, if  $L_{\infty}$  is estimated from length-based methods, as usually occurred in data-poor resources, and population exhibit signals of truncation (Saari *et al.* 2014), estimates of  $k$  will be equally biased because of the correlation between these two parameters (Beverton and Holt 1959). Given that estimates of  $M$  usually are obtained from  $L_{\infty}$  and  $k$ , which are relatively easier to estimate in comparison to  $M$  (Kenchington 2014), biases on these  $vBGP$  will be propagated in the estimates of  $M$ . Natural mortality was recognized in this thesis but also by other authors as the life-history parameter that produces the largest impact among the stock assessment outputs of fishery resources (Scott *et al.* 2016). For instance, a biased knowledge of  $M$  as a single value or related to  $k$  ( $M/k$ ) may be translated into a wrong definition of the stock status as well as produce a misleading view of the way to recover an overexploited resource (Gwinn *et al.* 2010). Length-based data is often used to estimate the  $vBGP$  and determine the stock structure (based on either length or age) that will be used in the stock assessment of data-poor resources. Therefore, both the biases in input data and the uncertainty in estimates of life-history parameters should be well recognized. Although the above-mentioned arguments make sense, stock assessment practice often overlook them by assuming (many times conveniently) that the input data are well known and out of biases. Given that stock assessment results are used to “recommend the future” of resources and people involved in harvesting them, the stock assessment should recover the scientific and social sense rather than, for example, be mainly focused on what is the next package that will be used and accepted by the stock-assessment working groups. Appropriate stock assessment of resources harvested by small-scale fisheries could not be achieved unless than the local community is constantly involved in data collection and local knowledge is used to get a less biased understanding of fishery history.



## 4. CONCLUSIONS



1. The pseudo-cohort virtual population analysis can offer reliable indicators on the population trends of data-limited resources (life-history parameters and catches are well known and unbiased), as shown for the case study of the Northwest Mediterranean European hake, yielding biases usually lower than 20% in comparison with conventional stock assessment of data-moderate resources based on standard VPA.
2. In order to interpret the temporal trend of the stock indicators that emerges from the pseudocohort VPA by using all sparse years together, it is recommendable to recognize the impact of variation of  $F_{ter}$  by year. Thus, this additional information helps to understand whether if the observed changes, for instance, on the stock size may be linked to real changes of the stock in the sea among years or are related to the guessed value of  $F_{ter}$  that was used.
3. The stock biomass of the European hake could have declined by half between the eighties and last assessed year (2014), while overfishing has remained.
4. The LB-SPR estimates for the European hake were not affected by the assumed growth hypothesis since  $M/k$  leads to similar values regardless of the used growth parameters. However, variation in model fits and consequently estimates of  $F/M$  and SPR depend on sample size and representation of the length structure distribution of the fished stock.
5. Uncertainty in life-history parameters of the European hake led to SPR estimates that were comparable to those derived from single “well-known” values (four life-history parameters sets individually used).
6. Fishing mortality will be overestimated and SPR underestimated by LB-SPR if exploitation is focused on juveniles while older individuals are partially inaccessible to fisheries.
7. The government-monitoring program in Colombia led to a misleading description of landings and fishing mortality. This triggered riskier definition of population status and of the overall fishery trend. The incorporation of data from the participatory monitoring program offered a better understanding of the exploitation patterns for the lane snapper and white grunt stocks.

8. Decentralization of fishery management structures and the participation of local communities are required to obtain improved data. If stock assessments produce a more reasonable population status for key species, reliable advice can be offered for participatory management of small-scale fisheries in data-poor contexts.
9. The uncertainty in the life-history parameters estimates may be addressed but is not easily reduced to the required level for providing an accurate SPR value. Thus, it should be preferred advising data-poor species as over- or underexploited rather than to strive for deciding what spawning potential ratio is left under the current fishing pressure.
10. According to this thesis findings, larger effort should be aimed at improving the direct estimates of the von Bertalanffy growth parameters and to obtain ageing confirmation through otolith -or other suitable hard structure- analysis. These procedures will allow improving the use of indirect methods for estimating natural mortality, and consequently, promoting a better definition of the stock status of data-poor species.
11. The stock assessment methods based on steady-state assumptions, which are used to determine the stock status of small-scale fisheries lacking knowledge of catch history, are capable to provide reliable estimates of the stock status trend. However, a proper status definition can only be achieved with a deep understanding of the biases underlying the input data, by means of sensitivity and uncertainty analyses that should be part of stock assessment routines.





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## 6. ANNEX



6.1. Annex 1. Supplementary  
material



## Supplementary material chapter 2.2

Table S1.  $F/M$  (variance) and SPR (variance) estimated from the fast-growth hypothesis ((Mellon-Duval et al. (2010), models 1 and 3) and slow-growth hypothesis by Aldebert and Recasens (1996) and the von Bertalanffy growth parameters estimated by this study.

Year	Mellon-Duval et al. (2010), model 3, M/k=3.17	Mellon-Duval et al. (2010), model 1, M/k=2.63	Aldebert and Recasens (1996), M/k=2.34	This study, M/k = 3.04	Mellon-Duval et al. (2010), model 3, M/k=3.17	Mellon-Duval et al. (2010), model 1, M/k=2.63	Aldebert and Recasens (1996), M/k=2.34	This study, M/k = 3.04
	$F/M$				SPR			
1998	2.27 (3.81E-03)	2.53 (4.39E-03)	2.97 (5.54E-03)	2.26 (3.76E-03)	0.03 (2.81E-06)	0.02 (2.03E-06)	0.02 (1.13E-06)	0.03 (2.96E-06)
1999	3.27 (6.94E-03)	3.61 (8.04E-03)	4.18 (1.01E-02)	3.25 (6.87E-03)	0.01 (5.11E-07)	0.01 (3.76E-07)	0.01 (2.10E-07)	0.01 (5.42E-07)
2000	3.97 (9.56E-03)	4.37 (1.11E-02)	5.03 (1.40E-02)	3.96 (9.44E-03)	0.01 (2.33E-07)	0.01 (1.75E-07)	0.00 (9.91E-08)	0.01 (2.48E-07)
2001	2.90 (9.42E-03)	3.21 (1.09E-02)	3.73 (1.38E-02)	2.88 (9.33E-03)	0.01 (1.54E-06)	0.01 (1.12E-06)	0.01 (6.26E-07)	0.01 (1.63E-06)
2002	2.07 (1.14E-02)	2.32 (1.33E-02)	2.73 (1.68E-02)	2.06 (1.13E-02)	0.03 (1.48E-05)	0.03 (1.07E-05)	0.02 (5.95E-06)	0.03 (1.56E-05)
2003	2.70 (8.50E-02)	3.01 (9.95E-03)	3.50 (1.25E-02)	2.69 (8.45E-03)	0.02 (2.18E-06)	0.01 (1.59E-06)	0.01 (8.82E-07)	0.02 (2.30E-06)
2004	3.26 (1.19E-02)	3.61 (1.39E-02)	4.17 (1.75E-02)	3.25 (1.18E-02)	0.01 (8.76E-07)	0.01 (6.43E-07)	0.01 (3.58E-07)	0.01 (9.28E-07)
2005	3.72 (2.44E-02)	4.11 (2.83E-02)	4.73 (3.57E-02)	3.71 (2.42E-02)	0.01 (7.08E-07)	0.01 (5.20E-07)	0.00 (2.91E-07)	0.01 (7.50E-07)
2006	3.44 (9.22E-03)	3.81 (1.08E-02)	4.40 (1.36E-02)	3.43 (9.16E-03)	0.01 (4.83E-07)	0.01 (3.55E-07)	0.01 (1.98E-07)	0.01 (5.11E-07)
2007	1.77 (1.22E-02)	1.97 (1.38E-02)	2.34 (1.75E-02)	1.75 (1.19E-02)	0.05 (4.11E-05)	0.04 (3.02E-05)	0.03 (1.70E-05)	0.05 (4.34E-05)
2008	2.07 (8.05E-03)	2.31 (9.30E-03)	2.72 (1.17E-02)	2.06 (7.96E-03)	0.03 (1.15E-05)	0.03 (8.51E-06)	0.02 (4.77E-06)	0.04 (1.22E-05)
2009	1.60 (5.13E-03)	1.82 (5.99E-03)	2.16 (7.55E-03)	1.60 (5.09E-03)	0.05 (2.51E-05)	0.04 (1.78E-05)	0.03 (9.80E-06)	0.05 (2.63E-05)
2010	1.40 (3.74E-03)	1.59 (4.31E-03)	1.90 (5.44E-03)	1.39 (3.70E-03)	0.08 (3.93E-05)	0.07 (2.86E-05)	0.05 (1.60E-05)	0.08 (4.14E-05)

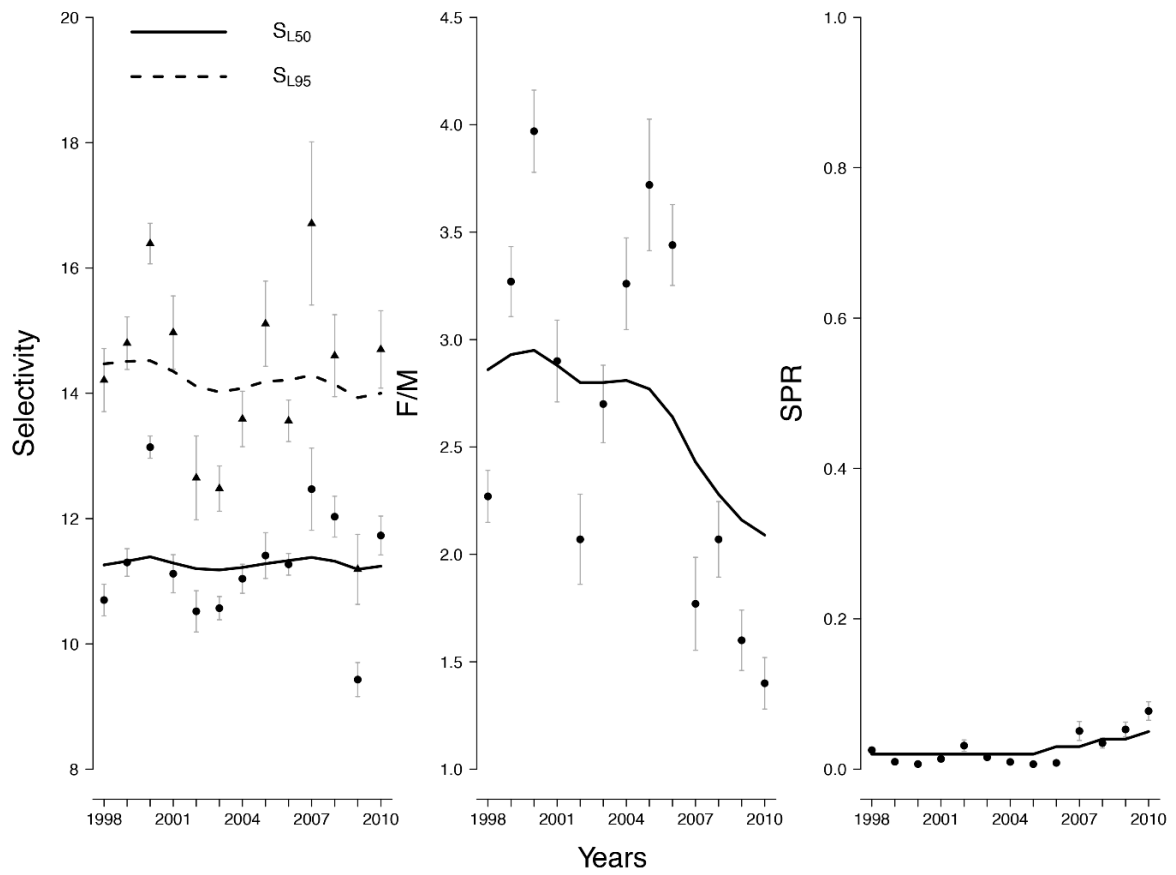


Figure S1. Annual (individual symbol) and smoothed model (black line) selectivity,  $F/M$  and  $SPR$  estimated by LB-SPR method from life-history parameters derived from Mellon-Duval et al., 2010, model 3 (fast-growth hypothesis). Variance is shown in vertical bars.

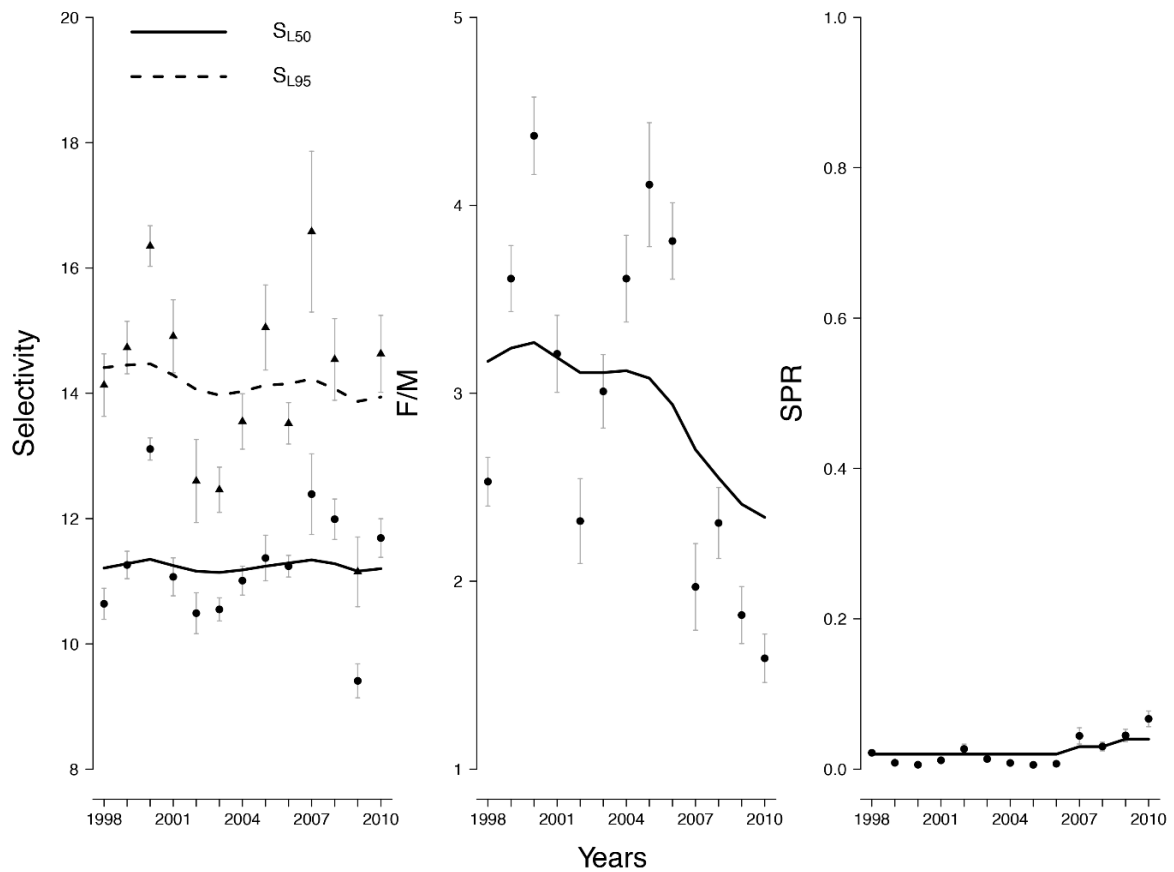


Figure S2. Annual (individual symbol) and smoothed model (black line) selectivity,  $F/M$  and  $SPR$  estimated by LB-SPR method from life-history parameters derived from Mellon-Duval et al., 2010, model 1 (fast-growth hypothesis). Variance is shown in vertical bars.

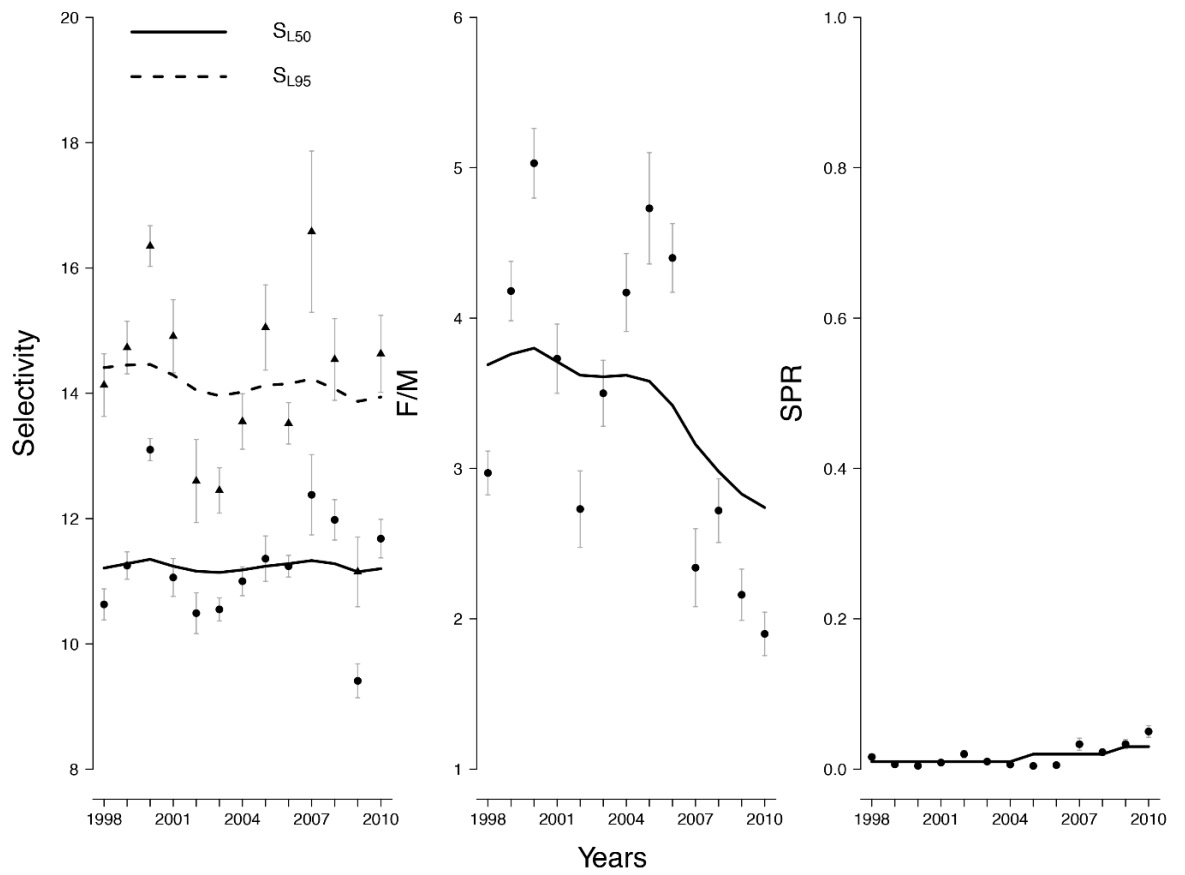


Figure S3. Annual (individual symbol) and smoothed model (black line) selectivity,  $F/M$  and  $SPR$  estimated by LB-SPR method from life-history parameters derived from Aldebert and Recasens, 1996 (slow-growth hypothesis). Variance is shown in vertical bars.

## Supplementary material chapter 2.3

Table S1. Length frequencies used to calculate the von Bertalanffy growth parameters of lane snapper. Data come from the participatory-monitoring program in the Central Guajira in 2010. Data was collected from fish caught by the bottom gillnet, encircling gillnet and longline fleets.

Total length (cm)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
13-14	0	0	2	0	0	0	0	0	0	0	0	0
14-15	0	0	0	0	0	0	0	0	0	0	0	0
15-16	0	0	0	0	0	0	2	0	0	0	0	0
16-17	0	2	0	0	0	0	0	0	4	0	0	2
17-18	0	0	2	0	0	2	0	2	2	0	0	0
18-19	0	0	0	0	2	0	0	2	8	0	0	0
19-20	0	0	0	0	0	2	2	2	2	0	0	0
20-21	2	0	0	0	0	2	0	4	4	0	0	2
21-22	4	4	0	0	0	2	0	4	6	2	10	2
22-23	2	0	0	4	0	8	4	6	10	0	2	4
23-24	8	8	4	0	4	6	2	0	16	2	12	6
24-25	0	12	6	14	4	12	2	6	14	2	6	12
25-26	8	25	2	10	14	16	6	2	12	4	16	8
26-27	10	14	8	23	8	21	10	14	16	2	23	8
27-28	12	8	21	25	23	14	19	10	16	2	25	6
28-29	6	4	29	14	21	6	4	8	12	4	19	2
29-30	21	6	12	19	19	10	19	12	10	10	29	4
30-31	10	0	31	14	4	8	21	14	14	8	6	4
31-32	14	0	19	12	8	2	14	6	23	6	2	2
32-33	8	0	4	8	2	6	10	14	16	6	8	0
33-34	12	0	8	8	4	4	4	6	6	4	4	0
34-35	8	0	4	8	4	6	2	4	12	2	2	0
35-36	6	0	4	4	4	2	2	2	4	2	8	0
36-37	4	0	2	4	4	2	2	2	6	0	4	0
37-38	6	0	6	2	0	0	2	0	8	0	0	0
38-39	4	2	2	6	4	4	0	0	2	0	0	2
39-40	6	0	2	0	2	2	4	0	2	0	0	6
40-41	2	0	0	2	0	0	0	0	2	0	6	4
41-42	2	0	0	0	0	0	2	0	0	0	0	2
42-43	0	0	0	0	0	0	0	0	0	0	0	0
43-44	2	0	0	0	0	0	0	0	2	0	0	0
44-45	0	0	0	0	0	0	0	0	2	0	0	0
45-46	0	0	0	0	0	0	0	0	0	0	4	0
46-47	0	0	0	0	0	0	0	0	0	0	2	0
47-48	0	0	0	0	0	0	0	0	2	0	0	0
48-49	0	0	0	0	0	0	2	0	0	0	0	0
49-50	0	0	0	0	0	2	0	0	0	0	0	0
50-51	0	0	0	0	0	0	0	0	0	0	0	0
51-52	0	0	0	0	0	0	0	0	0	0	0	0
52-53	0	0	0	0	0	0	0	0	0	0	0	0
53-54	0	0	0	0	0	0	0	0	0	0	0	0
54-55	0	0	0	0	0	0	0	0	0	0	0	0
55-56	0	0	0	0	0	0	2	0	0	0	0	0

## Supplementary material chapter 2.3

Table S2. Length frequencies used to calculate the von Bertalanffy growth parameters of lane snapper. Data come from the Government-monitoring program in the Central Guajira in 2013. Data was collected from fish caught by the encircling gillnet fleet.

Total length (cm)	Jan	Feb	Mar	May	Jun	Jul	Sep	Oct	Nov	Dec
19	1	0	0	0	0	0	0	0	0	0
20	4	0	1	1	13	1	1	0	1	0
21	33	3	4	15	13	2	8	1	6	2
22	54	4	4	19	23	8	14	9	7	1
23	82	1	0	32	13	6	8	16	16	6
24	85	2	2	27	17	5	13	21	20	13
25	50	1	4	24	10	3	15	17	13	17
26	40	0	3	29	5	0	2	18	16	16
27	17	0	0	24	7	2	3	12	9	44
28	22	0	1	19	4	0	3	3	5	14
29	17	0	2	14	2	0	0	1	4	7
30	9	0	0	3	2	0	1	1	3	2
31	6	0	0	3	0	0	0	4	5	1
32	4	0	1	2	0	0	3	0	3	0
33	4	0	1	7	1	2	0	0	3	0
34	5	0	0	3	0	0	0	1	1	0
35	1	0	0	1	0	0	0	1	2	0
36	1	0	0	1	0	0	0	0	2	0
37	2	0	1	0	0	0	1	0	2	0
38	1	0	0	0	0	0	3	0	1	0
39	0	0	0	0	0	0	1	0	1	0
40	1	0	0	0	0	0	1	0	1	0
41	0	0	0	0	0	0	1	0	1	0
42	1	0	0	0	0	0	0	0	0	0
43	0	0	0	0	0	0	0	0	0	1
44	0	0	0	0	0	0	0	0	0	1
45	0	0	0	0	0	0	0	0	0	1
46	0	0	0	0	0	0	0	0	0	0
47	0	0	0	0	0	0	0	0	0	0
48	0	0	0	0	0	0	0	0	0	1
49	0	0	0	0	0	0	0	0	0	0
50	0	0	0	0	0	0	0	0	0	0
51	0	0	0	0	0	0	0	0	0	0
52	0	0	0	0	0	0	0	0	0	0
53	0	0	0	0	0	0	0	0	0	0
54	0	0	0	0	0	0	0	0	0	0
55	0	0	1	0	0	0	0	0	0	0

## Supplementary material chapter 2.3

Table S3. Length frequencies used to calculate the von Bertalanffy growth parameters of white grunt. Data come from the participatory-monitoring program in the Central Guajira in 2010. Data was collected from fish caught by the bottom gillnet, encircling gillnet and longline fleets.

Total length (cm)	Ene	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
11-12	2	0	0	0	0	0	0	0	0	0	0	0
12-13	1	0	0	0	0	0	0	0	0	0	0	0
13-14	1	0	0	0	0	0	0	0	0	0	0	0
14-15	1	0	0	0	0	0	0	1	0	0	0	0
15-16	2	0	0	0	1	1	0	0	1	0	3	1
16-17	0	0	0	0	0	2	1	0	2	0	0	0
17-18	4	1	0	2	4	3	0	0	0	0	5	0
18-19	3	0	0	1	2	3	0	0	3	0	6	3
19-20	4	1	0	0	4	7	3	2	3	0	7	5
20-21	8	1	2	0	4	9	3	4	2	5	5	10
21-22	9	4	4	3	5	9	9	12	6	9	15	10
22-23	4	6	4	6	6	18	12	15	5	9	12	17
23-24	5	7	10	4	7	16	7	28	6	10	15	17
24-25	5	11	2	3	7	16	12	12	3	10	18	12
25-26	7	11	8	9	10	6	5	15	11	12	18	13
26-27	3	17	3	9	26	14	9	16	8	9	15	18
27-28	1	13	0	4	15	7	12	14	11	4	15	4
28-29	4	9	0	3	5	6	8	19	10	8	8	6
29-30	0	5	0	1	3	5	1	15	6	4	1	8
30-31	1	6	0	1	0	1	3	2	2	2	6	6
31-32	1	1	0	0	3	3	1	2	1	0	2	2
32-33	1	1	0	1	0	2	1	3	0	1	1	0
33-34	0	1	0	0	0	0	0	1	1	0	0	1
34-35	0	0	0	0	0	0	0	0	1	0	0	0
35-36	0	0	0	0	0	0	0	0	0	0	0	0
36-37	0	0	0	0	0	0	0	0	0	0	0	1
37-38	0	1	0	0	0	0	0	0	0	0	0	0
38-39	0	0	0	0	0	0	0	0	0	0	0	0
39-40	0	0	0	0	0	0	0	0	0	0	0	1

## Supplementary material chapter 2.3

Table S4. Vector of natural mortality calculated by Gislason's estimator.

	This study (2010)	This study (2013)	(Puentes et al., 2014)	This study (2010)	(Potts and Manooch, 2001)	(Dominguez-Viveros et al., 1996)
$L_{\infty}$	64.9	62.5	63.8	41.7	59.1	47.3
$K$	0.193	0.146	0.27	0.27	0.08	0.35
$t_0$	-1.576	-3.188	-0.661	-1.43	-4.21	-0.46
$a$	0.0173	0.0173	0.0173	0.008	0.008	0.008
$b$	2.917	2.917	2.917	3.172	3.172	3.172
Age	<i>M-at-age</i>					
0.50	0.45	0.40	0.56	0.46	0.69	0.68
1.00	0.24	0.20	0.30	0.23	0.36	0.35
1.50	0.17	0.13	0.22	0.15	0.25	0.24
2.00	0.13	0.10	0.17	0.11	0.19	0.19
2.50	0.11	0.08	0.15	0.09	0.16	0.15
3.00	0.09	0.07	0.13	0.07	0.14	0.13
3.50	0.08	0.06	0.12	0.06	0.12	0.12
4.00	0.08	0.05	0.11	0.06	0.11	0.10
4.50	0.07	0.04	0.10	0.05	0.10	0.09
5.00	0.06	0.04	0.10	0.04	0.09	0.09
5.50	0.06	0.03	0.09	0.04	0.09	0.08
6.00	0.06	0.03	0.09	0.04	0.08	0.08
6.50	0.06	0.03	0.08	0.03	0.08	0.07
7.00	0.05	0.03	0.08	0.03	0.07	0.07
7.50	0.05	0.02	0.08	0.03	0.07	0.07
8.00	0.05	0.02	0.08	0.03	0.07	0.06
8.50	0.05	0.02	0.07	0.03	0.07	0.06
9.00	0.05	0.02	0.07	0.02	0.06	0.06
9.50	0.04	0.02	0.07	0.02	0.06	0.06
10.00	0.04	0.02	0.07	0.02	0.06	0.05
10.50	0.04	0.02	0.07	0.02	0.06	0.05
11.00	0.04	0.02	0.07	0.02	0.06	0.05
11.50	0.04	0.02	0.07	0.02	0.06	0.05
12.00	0.04	0.01	0.07	0.02	0.05	0.05
12.50	0.04	0.01	0.06	0.02	0.05	0.05
13.00	0.04	0.01	0.06	0.02	0.05	0.05
13.50	0.04	0.01	0.06	0.02	0.05	0.05
14.00	0.04	0.01	0.06	0.01	0.05	0.04
14.50	0.04	0.01	0.06	0.01	0.05	0.04
15.00	0.04	0.01	0.06	0.01	0.05	0.04
15.50	0.04	0.01	0.06	0.01	0.05	0.04
16.00	0.04	0.01	0.06	0.01	0.05	0.04
16.50	0.03	0.01	0.06	0.01	0.05	0.04
17.00	0.03	0.01	0.06	0.01	0.05	0.04
17.50	0.03	0.01	0.06	0.01	0.05	0.04
18.00	0.03	0.01	0.06	0.01	0.05	0.04
18.50	0.03	0.01	0.06	0.01	0.04	0.04

## Supplementary material chapter 2.3

Table S5. Vector of natural mortality calculated by Abella's estimator.

	Lane snapper			White grunt		
	This study (2010)	This study (2013)	(Puentes et al., 2014)	This study (2010)	(Potts and Manoock, 2001)	(Dominguez-Viveros et al., 1996)
$L_{\infty}$	64.945	62.498	63.75	41.65	59.10	47.30
$K$	0.193	0.146	0.27	0.27	0.08	0.35
$t_0$	-1.576	-3.188	-0.66	-1.43	-4.21	-0.46
Length	<i>M-at-size</i>					
14	1.95	1.39	2.65	1.42	0.70	2.24
15	1.74	1.25	2.37	1.27	0.63	2.00
16	1.57	1.12	2.14	1.14	0.57	1.80
17	1.42	1.02	1.94	1.04	0.52	1.64
18	1.30	0.93	1.77	0.94	0.47	1.49
19	1.19	0.85	1.62	0.87	0.43	1.37
20	1.10	0.78	1.49	0.80	0.40	1.26
21	1.01	0.73	1.38	0.74	0.37	1.16
22	0.94	0.67	1.28	0.68	0.34	1.08
23	0.87	0.63	1.19	0.64	0.32	1.01
24	0.82	0.58	1.11	0.59	0.30	0.94
25	0.76	0.55	1.04	0.56	0.28	0.88
26	0.72	0.51	0.98	0.52	0.26	0.83
27	0.68	0.48	0.92	0.49	0.24	0.78
28	0.64	0.46	0.87	0.46	0.23	0.73
29	0.60	0.43	0.82	0.44	0.22	0.69
30	0.57	0.41	0.78	0.42	0.21	0.66
31	0.54	0.39	0.74	0.39	0.20	0.62
32	0.51	0.37	0.70	0.37	0.19	0.59
33	0.49	0.35	0.67	0.36	0.18	0.56
34	0.47	0.33	0.64	0.34	0.17	0.54
35	0.44	0.32	0.61	0.32	0.16	0.51
36	0.43	0.30	0.58	0.31	0.15	0.49
37	0.41	0.29	0.55	0.30	0.15	0.47
38	0.39	0.28	0.53	0.28	0.14	0.45
39	0.37	0.27	0.51	0.27	0.14	0.43
40	0.36	0.26	0.49			
41	0.34	0.25	0.47			
42	0.33	0.24	0.45			
43	0.32	0.23	0.44			
44	0.31	0.22	0.42			
45	0.30	0.21	0.40			
46	0.29	0.21	0.39			
47	0.28	0.20	0.38			
48	0.27	0.19	0.36			
49	0.26	0.19	0.35			
50	0.25	0.18	0.34			
51	0.24	0.17	0.33			
52	0.24	0.17	0.32			
53	0.23	0.16	0.31			
54	0.22	0.16	0.30			
55	0.21	0.15	0.29			
56	0.21	0.15	0.28			

## Supplementary material chapter 2.4

Table S1. Number of Peruvian grunt individuals gathered in the central coast of Peru from 2012 to 2015. The date corresponds to the central day of each campaign.

Total length (cm)	21/03/2012	13/11/2012	23/03/2013	14/11/2013	09/04/2014	27/10/2014	12/05/2015	06/11/2015
10.5	0	0	0	0	1	0	0	0
11.5	0	1	0	0	2	0	0	0
12.5	0	0	0	0	0	0	0	0
13.5	0	0	0	0	1	0	0	1
14.5	2	0	0	0	4	8	0	0
15.5	3	0	0	1	7	4	1	1
16.5	8	0	0	2	8	3	0	2
17.5	6	0	2	1	19	4	3	2
18.5	7	4	3	4	16	6	2	2
19.5	12	7	5	5	12	13	4	6
20.5	13	9	4	12	18	17	5	15
21.5	13	6	4	16	15	14	6	14
22.5	23	18	7	19	22	17	8	12
23.5	20	8	5	30	24	19	5	22
24.5	22	14	4	17	35	13	9	23
25.5	24	9	2	21	51	14	16	23
26.5	11	13	13	38	32	22	35	24
27.5	13	26	13	50	60	20	18	30
28.5	19	18	10	65	57	19	12	32
29.5	9	23	23	42	46	15	13	36
30.5	7	21	13	39	48	21	12	28
31.5	8	18	8	31	29	6	4	20
32.5	6	7	8	30	43	9	2	29
33.5	2	11	2	15	34	2	2	18
34.5	2	10	9	17	31	5	2	15
35.5	5	4	7	8	19	2	2	6
36.5	2	6	4	5	23	5	5	4
37.5	2	1	3	4	18	1	5	6
38.5	1	4	2	5	17	2	2	2
39.5	1	4	1	3	15	0	2	1
40.5	0	1	1	3	9	2	2	3
41.5	2	1	3	1	6	0	3	3
42.5	0	0	3	2	6	1	1	1
43.5	0	1	2	0	5	1	0	0
44.5	1	0	1	3	4	0	1	3
45.5	1	2	1	0	3	0	0	0
46.5	0	1	2	0	1	0	1	1
47.5	0	2	2	0	2	0	0	1
48.5	0	0	1	0	2	1	0	3
49.5	0	0	1	1	2	0	0	0
50.5	0	0	1	0	2	0	0	0
51.5	0	0	0	0	5	0	0	0
52.5	1	1	1	0	0	0	0	0
53.5	1	0	0	0	0	0	0	0
54.5	0	0	0	0	0	0	0	0
55.5	0	0	2	0	1	0	0	0

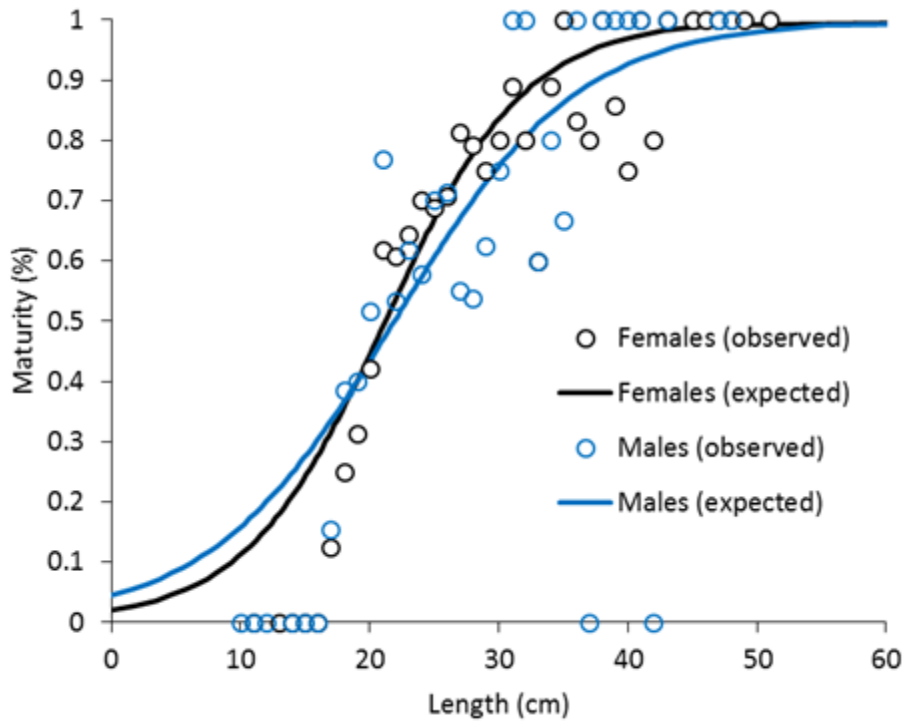


Figure S1. Maturity ogives used to estimate  $L_{50}$  and  $L_{95}$  of female (black) and male (blue) of the Peruvian grunt stock in the center coast of Peru.

## Supplementary material chapter 2.4

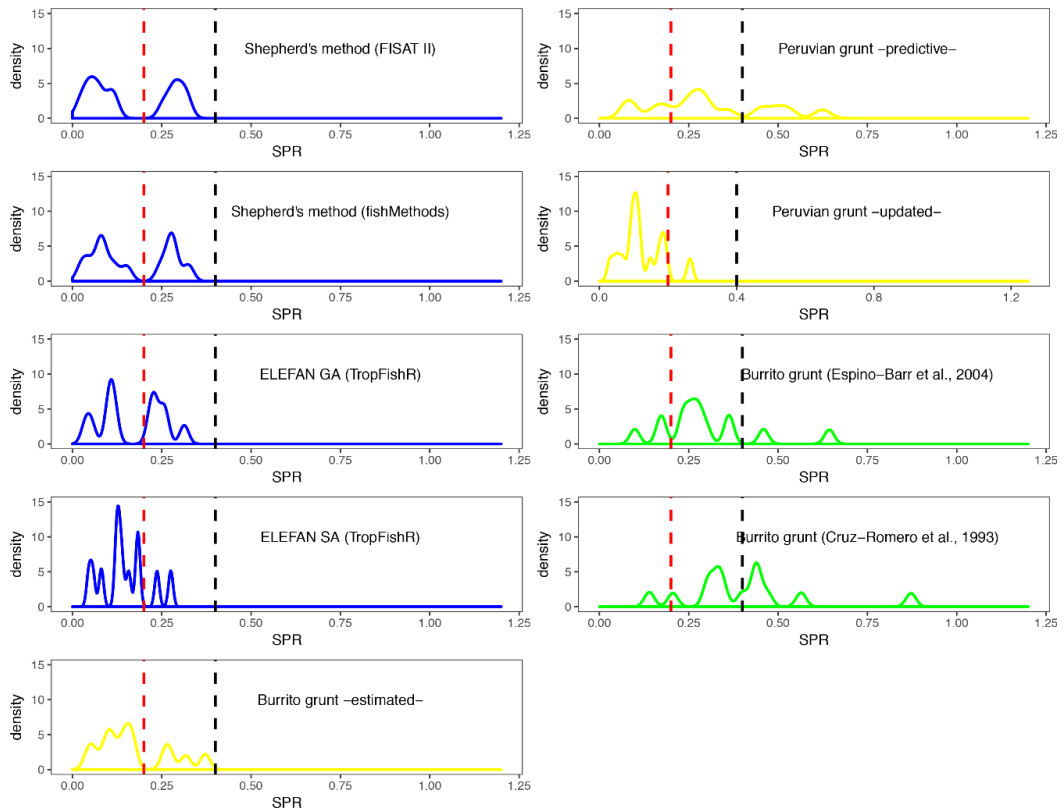


Figure S2. SPR distribution in 2012 produced by proportional contribution of the life-history parameters derived from the nine different methods for estimating the *vBGP*. Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

## Supplementary material chapter 2.4

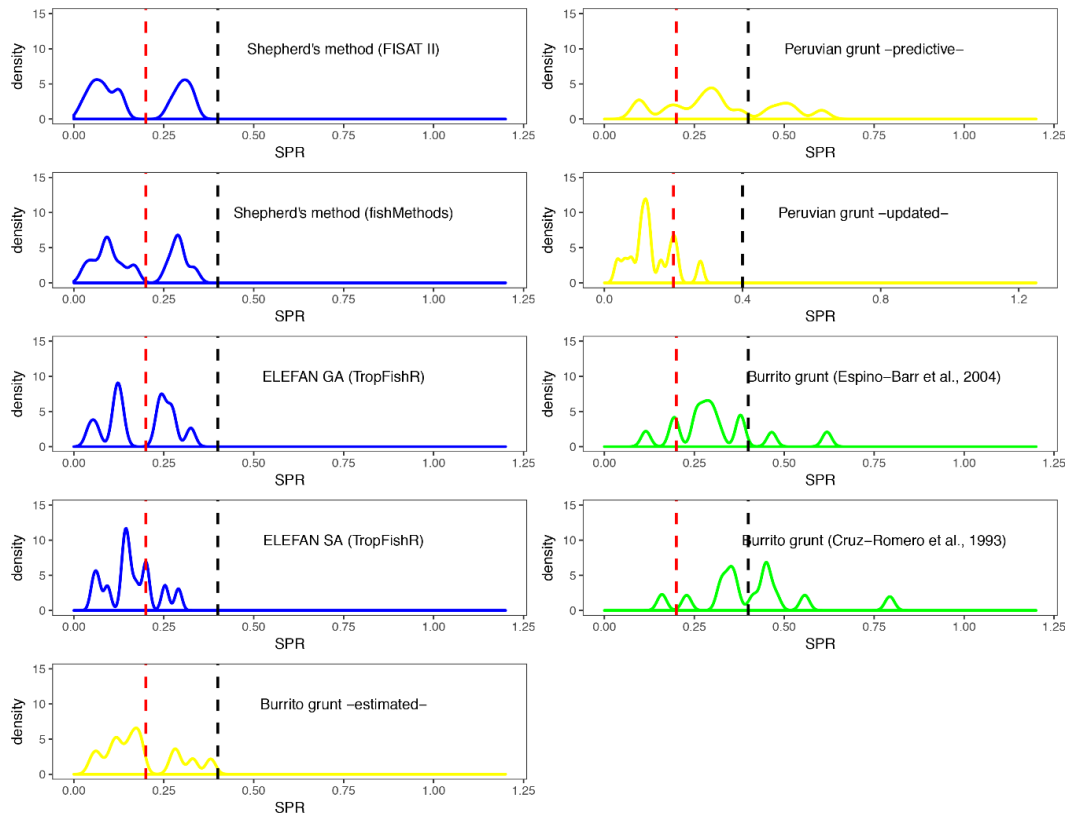


Figure S3. SPR distribution in 2013 produced by proportional contribution of the life-history parameters derived from the nine different methods for estimating the  $vBGP$ . Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

## Supplementary material chapter 2.4

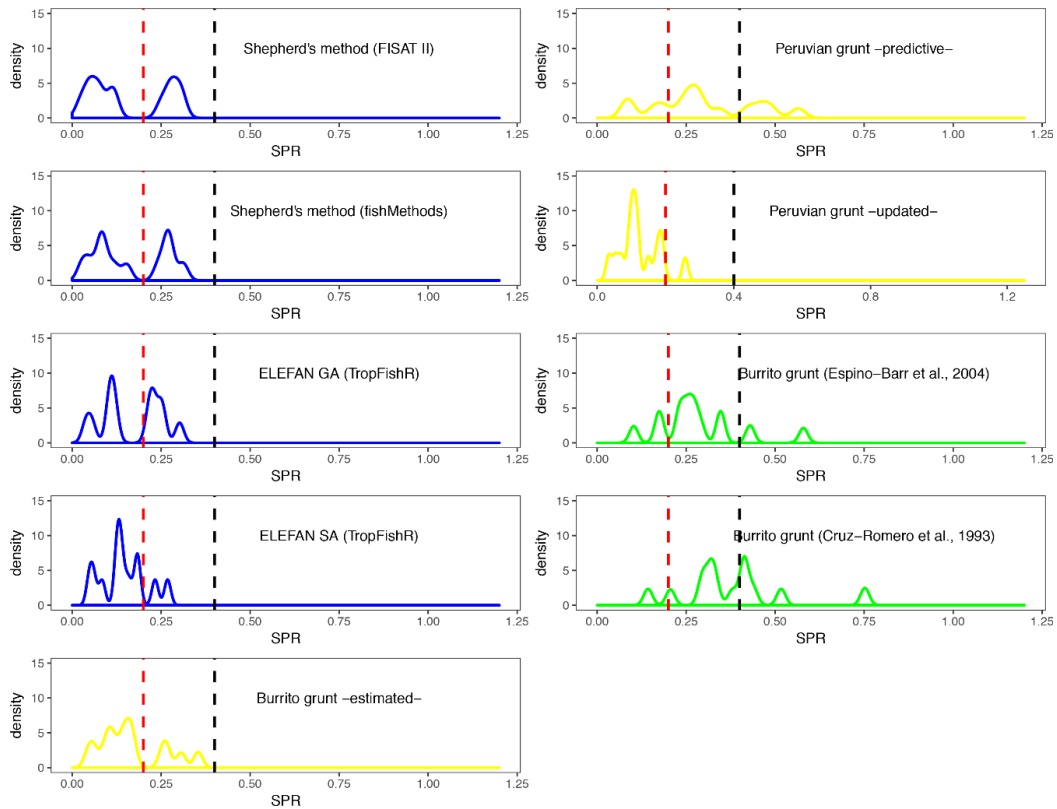


Figure S4. SPR distribution in 2015 produced by proportional contribution of the life-history parameters derived from the nine different methods for estimating the  $vBGP$ . Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

## Supplementary material chapter 2.4

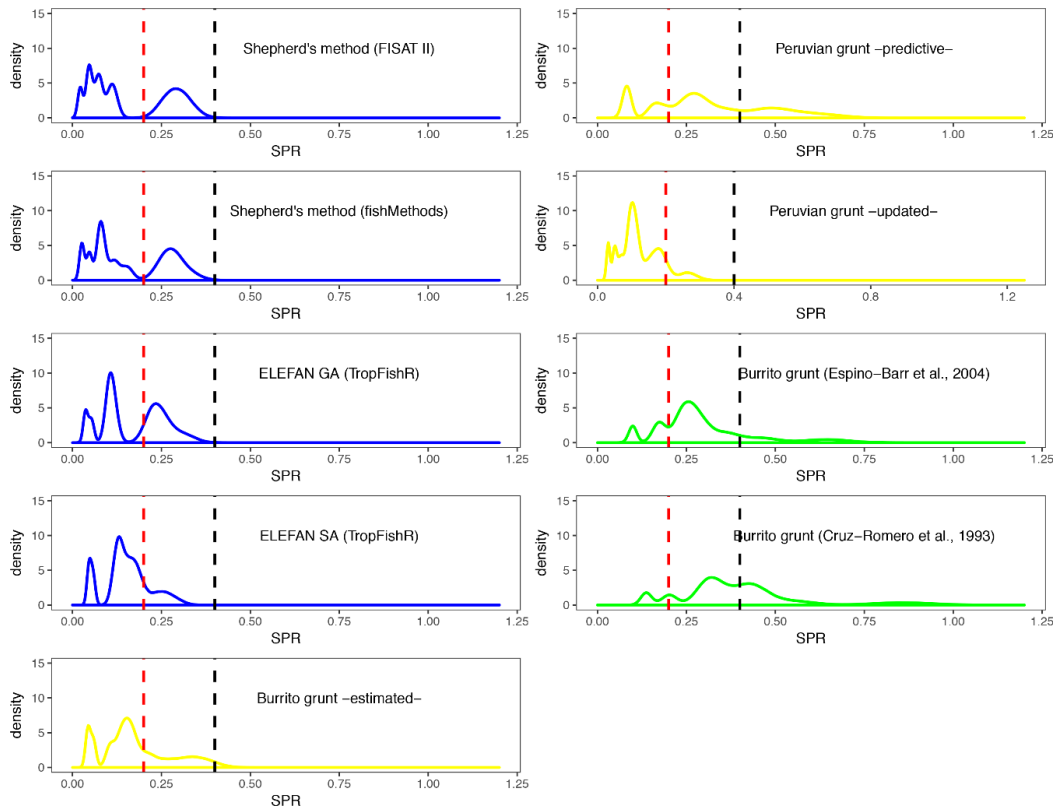


Figure S5. SPR distribution in 2012 obtained from the life-history parameters distribution uncertainty estimated by the Monte Carlo simulation. Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

## Supplementary material chapter 2.4

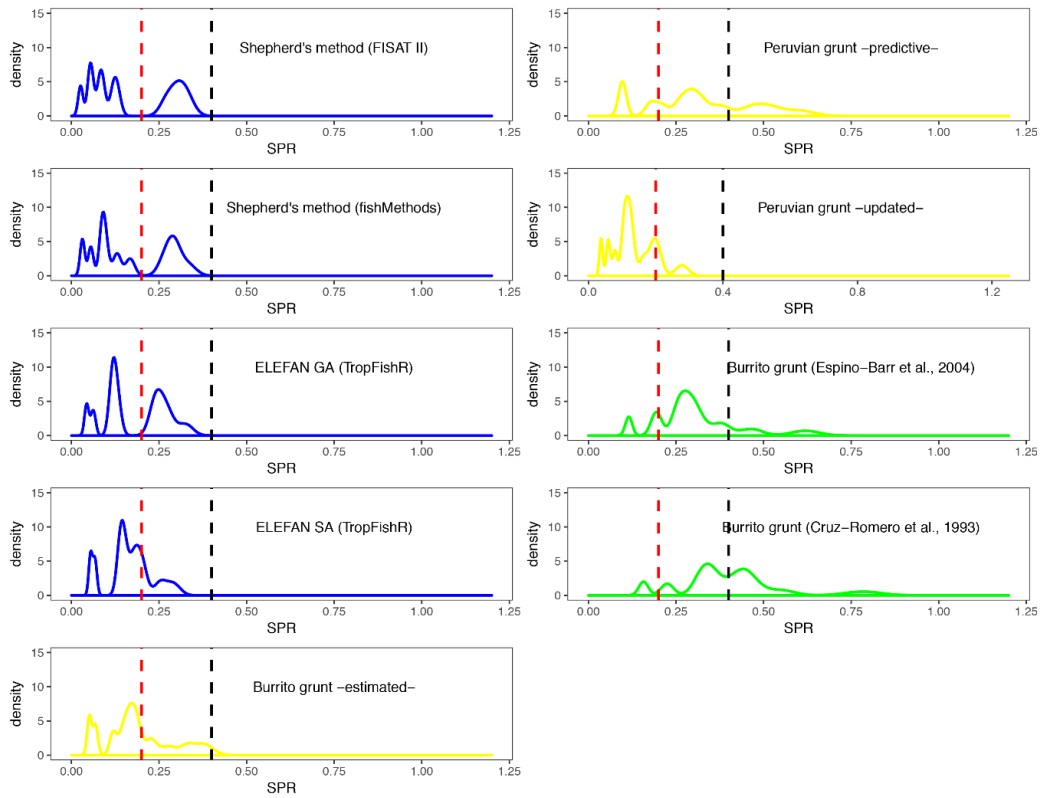


Figure S6. SPR distribution in 2013 obtained from the life-history parameters distribution uncertainty estimated by the Monte Carlo simulation. Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.

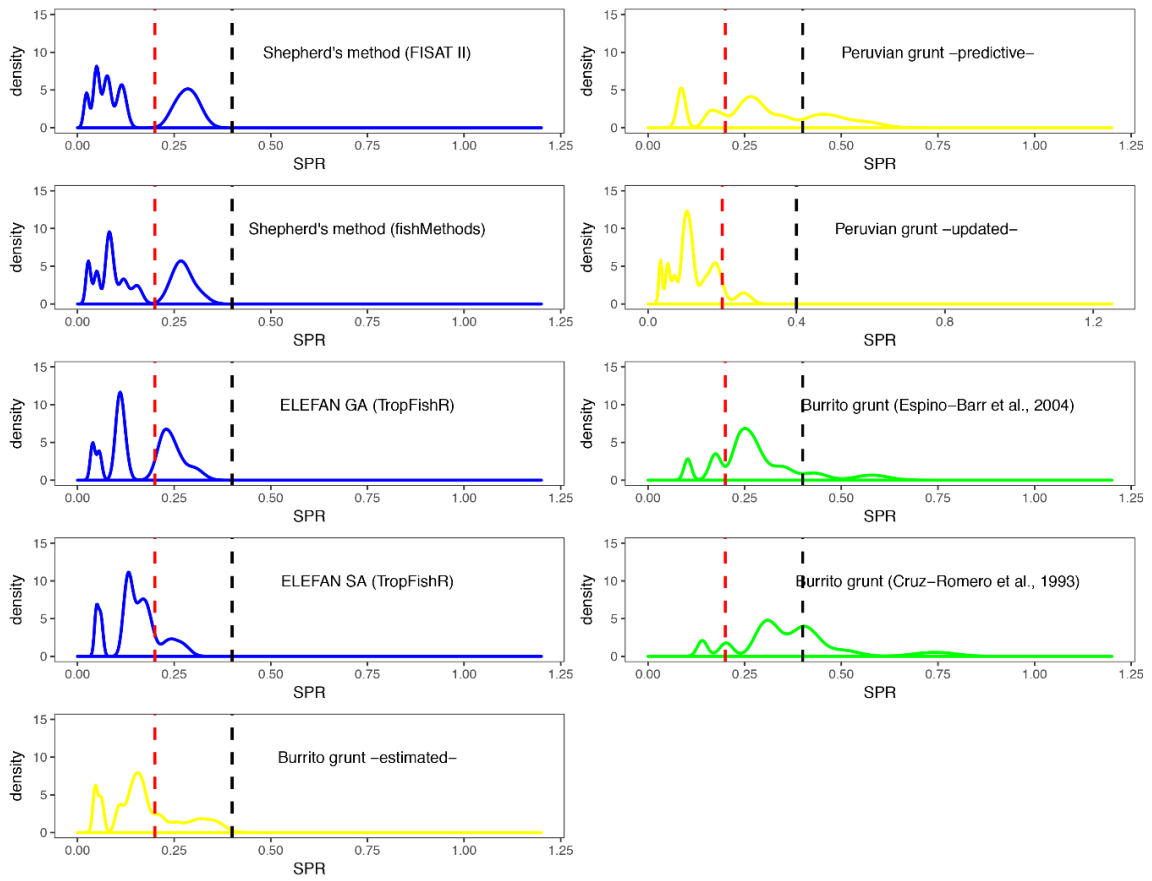


Figure S7. SPR distribution in 2015 obtained from the life-history parameters distribution uncertainty estimated by the Monte Carlo simulation. Length-based (blue lines) and *FishLife* (yellow lines) methods as well as an extent of “Robin Hood” approach (green lines) are plotted. Reference points of  $SPR_{0.2}$  (red line) and  $SPR_{0.4}$  (black line) are indicated.



## 6.2. Annex 2. Published papers

This annex section presents the papers published for chapters 2.2 and 2.3





Contents lists available at ScienceDirect

Fisheries Research

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## European hake (*Merluccius merluccius*) assessment based on size frequencies and basic biological parameters in the SW Mediterranean



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### ABSTRACT

Assessments of small-scale, coastal fisheries are often hindered by the lack of complete data sets fulfilling the requirements of conventional stock assessment methods. In contrast, existing information from the regular monitoring of fisheries coupled with data-limited assessment methods can help provide the information needed for effective management of fisheries. We investigated the year-to-year length-based estimates of the spawning potential ratio (SPR) of European hake, a species with moderate data availability (i.e., reasonable knowledge of the life-history traits and temporal length structures of the population). We explore the effects of considering fast- vs. slow-growth hypotheses and the uncertainties of input parameters on the annual estimates of the SPR (1998–2010) in a bottom trawl fishery in the SW Mediterranean (Murcia region). The SPR estimates under four life-history scenarios were sensitive to the quality (sample size) of the length frequencies. Consequently, comparable and more accurate SPR estimates were often found for annual samples larger than 2000 individuals. The  $F/M$  ratio varied less when size compositions were grouped by period, suggesting there are transitory-population size structures. The SPR estimates were insensitive to the underlying growth hypothesis since similar  $M/k$  ratios  $> 2.3$  were derived from the four sets of von Bertalanffy growth parameters. The assessment indicated overexploitation of hake ( $F/M > 2$ ,  $SPR < 10\%$ ) that is comparable with that derived from conventional assessments for this species elsewhere in the Mediterranean Sea. According to our findings, the LB-SPR method can provide reliable stock assessments and allows population trends to be calculated for data-limited species. However, this method requires a good understanding of the input data and their possible sources of bias.

### 1. Introduction

Stock statuses of target fishery resources have conventionally been estimated using virtual population analyses (Hilborn and Walters, 1992; Lassen and Medley, 2001) and statistical catch-at-age models (Methot and Wetzel, 2013; Scott et al., 2016). Information on life-history parameters is consequently expected to be reasonably well-known, and the volume of landings and size distribution of catches are expected to be properly monitored at least twice during the species lifespan. Fishery-independent data are also used to provide abundance/biomass indices. The fishery resources that meet these requirements are commonly categorized as “data-rich” species (Kokkalis et al., 2015).

However, many target species, as well as most stocks in multispecies fisheries, do not meet the requirements for assessment using data-rich methods. These species, by default, are usually considered data limited. The assumptions underlying the methods to assess data-limited fisheries can circumvent the lack of fishery information (e.g., catch-at-age).

However, defining, for example, how much the fishing mortality is above or below a reference point requires appropriate knowledge of life-history traits (Kokkalis et al., 2015; Magnusson and Hilborn, 2007; Ramírez et al., 2017).

The spawning potential ratio (SPR; the ratio of the reproductive potential of a stock subject to a given amount of fishing pressure to that of an unfished stock; Goodyear, 1993) can be estimated from length-based techniques using data-limited methods (Hordyk et al., 2015c). SPR ranges between 0 and 1, with a value of 1 representing an unexploited stock. Satisfactory snapshots of SPR estimates have been obtained for species with little or no knowledge of their life histories (Prince, 2017; Prince et al., 2015). The performance of this method depends on the accurate estimates of life history ratios, and it is especially sensitive to the estimates of  $L_{\infty}$  (Hordyk et al., 2015c). In addition, Prince et al. (2015) found that the most uncertain assessments were derived from small size samples. Accordingly, the uncertainties of length-based spawning potential ratio (LB-SPR) outputs were quantified

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by these authors by considering plausible values of life-history parameters and resampling the available length data.

Hake (*Merluccius merluccius*) is one of the most important European target species of bottom trawl nets and set longlines, and annual catches of ca. 25,000 t have been reported in recent years from Mediterranean waters (FAO, 2011). According to conventional stock assessments, this species is overexploited in all Mediterranean fishing subareas in recent decades (Cardinale and Scarcella, 2017; Colloca et al., 2013). However, unsustainable hake harvest was already a concern before that, at least for northwest Mediterranean stocks (Aldebert et al., 1993; Bas et al., 1955; Oliver, 1994).

The stock assessments of hake in the northwest and southwest Mediterranean Sea by working groups of the STECF (Scientific, Technical and Economic Committee for Fisheries) or the GFCM (General Fisheries Commission for the Mediterranean) in recent years have relied on the fast-growth hypothesis that was principally justified from a tag-recapture experiment in the Gulf of Lions (Mellon-Duval et al., 2010) which determined two different growth curves when females (model 1) or females, males and undetermined sexes (model 3) were considered. On the other hand, the slow-growth hypothesis of hake was considered to be appropriate in early assessments (Aldebert and Recasens, 1996) and is supported by this study (see below). Consequently, we examined the sensitivity of the results of the LB-SPR method to a range of  $L_{\infty}$  and  $k$  values from 100.7 to 110 cm and from 0.112 to 0.236  $\text{yr}^{-1}$ , respectively (“slow” and “fast” growth hypotheses).

Reasonable bounding estimates of the von Bertalanffy growth parameters ( $vBGP$ ) and natural mortality ( $M$ ) of the hake stock in the Mediterranean Sea and a comprehensive set of length frequencies recorded onboard a commercial fleet from 1998 to 2010 in the Murcia region (southwest Mediterranean Sea) provide high-quality data for estimating year-to-year length-based SPR values. In this sense, hake will be considered as a species with moderate data, assessed using a data-limited framework. Therefore, the objective of this work is to assess the exploitation rate of hake in the SW Mediterranean based on a size-frequency analysis of individuals sampled from commercial catches over 13 years using the LB-SPR method under different assumptions and uncertainties of life-history parameters. We also examine whether the results from the LB-SPT method deviate from those of conventional assessment methods.

## 2. Materials and methods

In the Murcia region, as elsewhere in the Mediterranean, hake is of high economic importance for the trawl fleet. Catches for 1998–2010 fluctuated from 208 t in 2000 to 60 t in 2008, with an average catch of 109 t (Table 1). The trawl fleet has declined continuously in number from 39 in 2000 to 29 in 2010. The earliest formal stock assessment in the Murcia region, corresponding to data between July 1998 and June

**Table 1**  
Landings (tonnes) and numbers of measured individual hake fished by trawlers during 1998–2010 in the Murcia region (SW Mediterranean, Spain).

Year	Landings (t)	Total individuals
1998	111.670	4436
1999	102.541	4505
2000	208.230	4199
2001	184.340	2697
2002	123.334	1041
2003	83.942	1990
2004	93.973	2231
2005	66.903	1691
2006	109.724	2985
2007	89.571	1416
2008	59.992	1608
2009	109.105	1594
2010	76.100	1948

1999, showed strong overexploitation of the species (Martínez-Baños, 2000), which has continued, as evidenced in the recent stock assessments of subarea 1 by the GCFM, to which the Murcia region belongs (STECF, 2015, 2011).

### 2.1. Length frequency data

Hake, along with other demersal species targets of the trawl fishery, are regularly monitored in the Murcia region by the local fisheries department of the Autonomous Government, in the ports of Mazarrón, Águilas and Cartagena (Fig. 1).

Hake were measured onboard commercial trawlers working on hake fishing grounds (50–350 m depth approximately) in the Murcia region (Fig. 1) from April to December 1998 to 2010, from 85 fishing trips. Two or three fishing trips were sampled each month over three depth strata: stratum A 50–150 m, stratum B 150 and 350 m, and stratum C deeper than 350 m. The sampled data were weighted by catch-in-number by depth stratum and month to obtain a representative annual size distribution. Thus, the contribution of the sampled individuals to the size structure of the stock caught by the bottom trawl was proportional to the total individuals harvested by the fleet every year.

Fish brought onboard were randomly sampled across all commercial categories (small, medium and large). Individual sizes (cm TL) were measured to the lowest half centimetre. We measured 31,969 hake with sizes ranging between 5 and 74 cm TL. The modal value was 16.5 cm TL, and 95% of individuals were between 7 and 37.5 cm TL. Length frequency data after 2010 were not considered because of regulatory mesh changes in Mediterranean trawls (adoption of 40-mm square meshes) that could imply changes in catchability. Hake is not fished with fishing gear other than bottom trawls in the Murcia region (i.e., set longlines or gillnets are not employed).

Despite the fact that the trawl gear is often assumed to catch a wide range of sizes, some studies have indicated a dome-shaped selectivity pattern (Binion et al., 2009; Kuparinen et al., 2009). This pattern occurs because the size distribution perceived by trawl samples may be caused by ontogenetic migrations to deep waters (Jacobson et al., 2001). In particular, this behaviour has been associated with the European hake in the West Mediterranean (Abella et al., 1997; Caddy, 2015). In addition, overrepresentation of some sizes in trawl catches may occur depending on the growth hypothesis for the species (Huse et al., 1999). Consequently, the asymptotic selectivity assumed by the LB-SPR model (Prince et al., 2015) may not be totally valid, requiring careful interpretation of the SPR results for European hake.

### 2.2. Life-history parameters

We used four sets of von Bertalanffy growth parameters, three taken from the literature and the fourth estimated in this study from the raised annual size frequency. There are two contrasting hypotheses regarding hake growth in the Mediterranean, with  $k$  values of approximately 0.1  $\text{yr}^{-1}$  for the slow-growth hypothesis and  $k$  values of approximately 0.2  $\text{yr}^{-1}$  for the fast-growth hypothesis (Table 2). We used models 1 and 3 of Mellon-Duval et al. (2010), which were derived for a hake stock in the Gulf of Lions as representatives of the “fast growth” hypothesis. The two parameter sets representing the “slow growth” hypothesis (Table 2) were obtained from Aldebert and Recasens (1996, females) and from the length frequencies used in this study (Martínez-Baños et al. (Data in Brief)), pooling all years and using the MIX model (Macdonald and Green, 1988; software MIX R version: library *mixdist* v. 0.5–4 in R v. 3.0.2). The results obtained for the size-age distribution of hake in the Murcia region were  $L_{\infty} = 105.874$  cm TL,  $k = 0.112$   $\text{yr}^{-1}$  and  $t_0 = -0.167$  yr.

The LB-SPR method assumes constant natural mortality ( $M$ ) for all recruited size classes. However, the stock assessment of hake in the Mediterranean Sea uses an  $M$ -at-age vector (1.24, 0.58, 0.45, 0.40, and 0.37 for ages 0–5+) (STECF, 2015), which was estimated using the

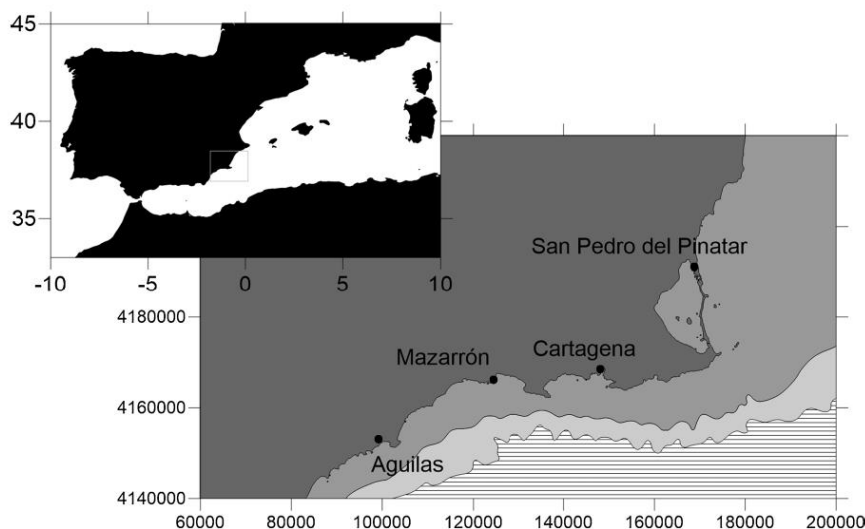


Fig. 1. Study area (Murcia Region) showing harbours where the bottom trawl fleet landings were monitored. Depth contours shown are 200 and 1000 m.

PRODBIOM indirect estimator (Abella et al., 1997). This estimator is frequently used to estimate  $M$  for Mediterranean resources. In particular, this  $M$  estimator supports the “refuge” concept for older hake ages, which was considered by Abella et al. (1997) as being partially responsible for the sustained productivity of fisheries harvesting the juvenile fraction of stocks. Given that PRODBIOM uses values for  $L_{\infty}$  and  $k$ , along with other information to estimate  $M$ , each set of von Bertalanffy growth parameters produced a different  $M$ . Model 3 (Mellon-Duval et al., 2010) is used in the official stock assessment (STECF, 2011). The  $M$  derived from the abovementioned estimator met the considerations made by ICES (2014), who advised that a fast-growth hypothesis for hake will imply an  $M$  value that is nearly double that implied by the slow-growth hypothesis (Table 2).

### 2.3. LB-SPR method

The method developed by Hordyk et al. (2015a) calculates SPR for a stock directly from the size composition of the catch. It requires the size composition data of the catch as input data as well as a minimum set of biological parameters:  $L_{\infty}$ , the coefficient of variation (CV) of  $L_{\infty}$ ,  $M/k$  and the size of maturity. The four  $L_{\infty}$  parameters and the ratios between natural mortality and the von Bertalanffy growth rate ( $M/k$ ) were estimated (Table 2).

Length observations were abundant each year ( $n > 1000$ , Table 1), but ageing was not performed. According to Prince et al. (2015), the CV of  $L_{\infty}$  can be assumed to be approximately 10% when reliable length and age data are not available. We assumed that this value was valid to including in the LB-SPR model for European hake. In addition, we

explored the sensitivity to this value by considering CVs for  $L_{\infty}$  of 0.05, 0.1, 0.15, 0.2, 0.25 and 0.3, compared to a hypothetical low-variance value of CV of  $L_{\infty} = 0.001$  (assumed “true” value). A boxplot was used to show the LB-SPR outputs ( $F/M$  and SPR) from 52 runs for each assumed value of the CV of  $L_{\infty}$ .

The sizes at which 50% ( $L_{50}$  = 30 cm) and 95% ( $L_{95}$  = 40 cm) of the population is mature were obtained from the raw data used for the official stock assessment of hake (STECF, 2015). From the life-history ratios ( $M/k$ ,  $L_{50}/L_{\infty}$  and  $L_{95}/L_{\infty}$ ), the model allows the estimation of the selectivity at length ( $L_{S_{50}}$  and  $L_{S_{95}}$ ) and the exploitation ratio  $F/M$ , which are used to calculate the SPR (see Hordyk et al., 2015c for a full description of the model).

### 2.4. Uncertainty of life-history parameters

Additional uncertainties in the SPR outputs were quantified by placing distributions bounded by upper and lower values of  $L_{\infty}$ , the CV of  $L_{\infty}$ ,  $M/k$ ,  $L_{50}$  and  $L_{95}$  to account for the uncertainties of these parameters. In addition to the R library, “LBSR” used to estimate the SPR for each year, the R library “triangle” was used to generate the life-history parameters.

The upper value of the mean asymptotic length ( $L_{\infty}$ ) was 115 cm because the hake population in the Murcia region, according to the current evidence, is largely truncated. A maximum value of 110 cm could underestimate  $L_{\infty}$  for an underexploited population. We selected 105 cm as the mean  $L_{\infty}$  value, given that this value corresponded to estimates from the Murcia region data (the lower  $L_{\infty}$  was 95 cm). The lower, upper and middle values of the triangular distribution for the CV

Table 2

von Bertalanffy growth parameters and natural mortality of hake in the Mediterranean Sea used in this study.  $M$  was estimated using PRODBIOM from the average  $M$  at ages 0–5 (Abella et al., 1997).

Source	Region	Sex	$L_{\infty}$ (cm)	$M$ ( $\text{yr}^{-1}$ )	$K$ ( $\text{yr}^{-1}$ )	Growth hypothesis	$M/k$
Mellon-Duval et al. (2010), model 3	Gulf of Lions	Male, female, undetermined	110	0.565	0.178	Fast	3.17
Mellon-Duval et al. (2010), model 1	Gulf of Lions	Female	100.7	0.62	0.236	Fast	2.63
Aldebert and Recasens (1996)	Catalan Sea	Female	100.7	0.29	0.124	Slow	2.34
Martínez-Baños et al. (Data in Brief)	Murcia	Combined	105.87	0.34	0.112	Slow	3.04

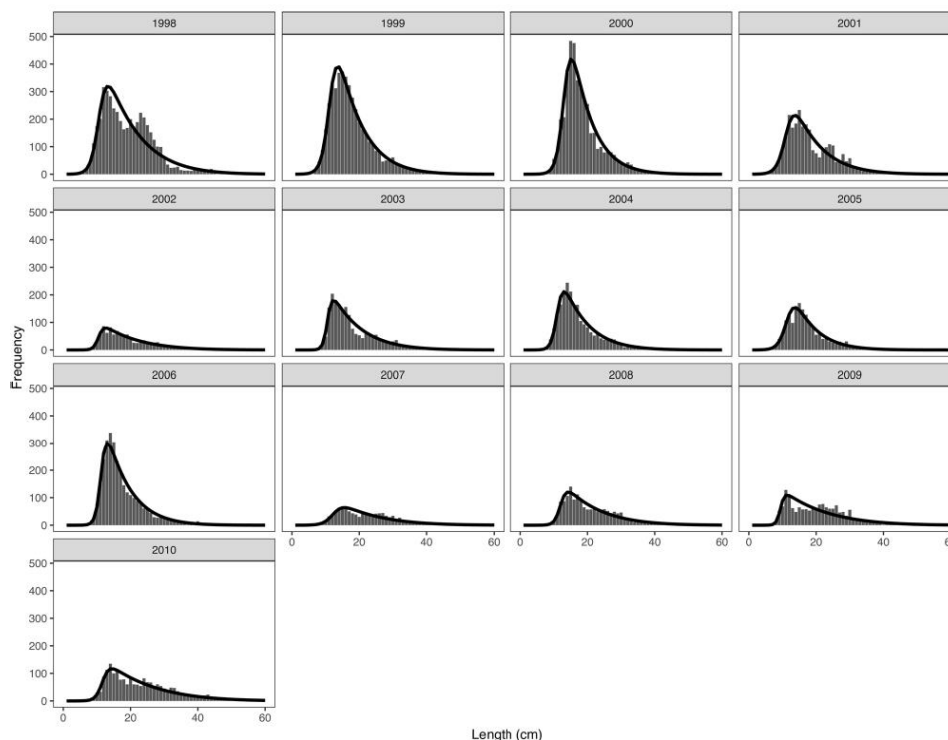


Fig. 2. Observed and LB-SPR model-predicted length-frequency histograms (bars and gray lines respectively) when the LB-SPR model was fitted using the life-history parameters derived from the Murcia region.

of  $L_{\infty}$  were set to 0.05, 0.3 and 0.15, respectively. Considering the life-history parameters used in this study, we used  $M/k$  values from 2.2 to 3.2 (mean  $M/k = 2.75$ ). Given that the estimation of the asymptotic length of hake may be subject to considerable uncertainty, but  $L_{50}$  is comparatively better known, we assumed a constant  $L_{50}/L_{\infty}$  ratio = 0.28. We used 10,000 random draws to obtain the distributions of the life-history parameters and the final distribution of the LB-SPR estimates per year.

The population may be far from equilibrium, implying that size composition is affected by the transitory dynamics of mortality and recruitment. Assuming that the length frequency data are the only available information, we explored the  $F/M$  and SPR estimates by grouping the annual length frequencies in four periods (1998–2000, 2001–2003, 2004–2006 and 2007–2010). Thus, we provided a smoothed trend of the LB-SPR estimates to reduce the potential effect of the annual non-equilibrium state.

### 3. Results

Fig. 2 shows the fit to the length frequency data by year for the growth parameters estimated in this study. For this set of parameters,  $L_{50}$  varied around 11.5 cm TL and  $L_{95}$  around 14 cm TL (Fig. 3 left panel) while  $F/M$  decreased substantially from ca. 3.0 in 2007 to ca. 2.0 in the most recent years (Fig. 3 central panel), and the SPR was estimated to less than 0.1 but to have increased in the most recent four years (Fig. 3 right panel). The model outputs were similar for the other

three life-history parameter sets in Table 2 (Figs. S1, S2 and S3). The selectivity parameters remained quite constant (average  $L_{50} \sim 11.2$ ;  $L_{95} \sim 14.2$  across the four growth curves, Figs. 3, S1, S2 and S3), i.e. the values of the selectivity parameters were insensitive to the life-history ratios used as inputs to the LB-SPR model (CV < 0.5% for both  $L_{50}$  and  $L_{95}$ , not shown).

The highest ratio between fishing and natural mortality  $F/M$  (Fig. 4, Table S1) was for the lowest  $M/k$  value (2.34). Conversely, high  $M/k$  values (3.04 and 3.17), regardless of the growth hypothesis, led to the lowest estimates of  $F/M$ .

Although mean  $F/M$  derived from the four life-history parameters varied by  $\sim 79\%$  per year from 1998 to 2010, the corresponding SPR estimates exhibited limited changes during some years (e.g., 1999, 2004 and 2006, Fig. 4, Table S1) or were clearly differentiated in others (e.g., 2007–2010, Fig. 4, Table S1). Fig. 2 indicates that the LB-SPR model was able to fit the length-frequency distributions for 1999, 2004 and 2006 better than those for other years, with the best fits coinciding with large samples sizes ( $n > 2000$ ) (Table 1). Conversely, many of the annual models exhibiting poor fits to the length-frequency distributions (2007–2010, Fig. 2) coincided with sample sizes less than 2000 individuals (Table 1). Thus, a good fit to the length frequency data by year usually led to similar SPR values (Fig. 4, Table S1), while poor fits tended to produce noticeable differences in the SPR estimates. We speculate that a larger number of length samples each year contributed to reducing the differences between the SPR estimates derived from the four life-history parameters.

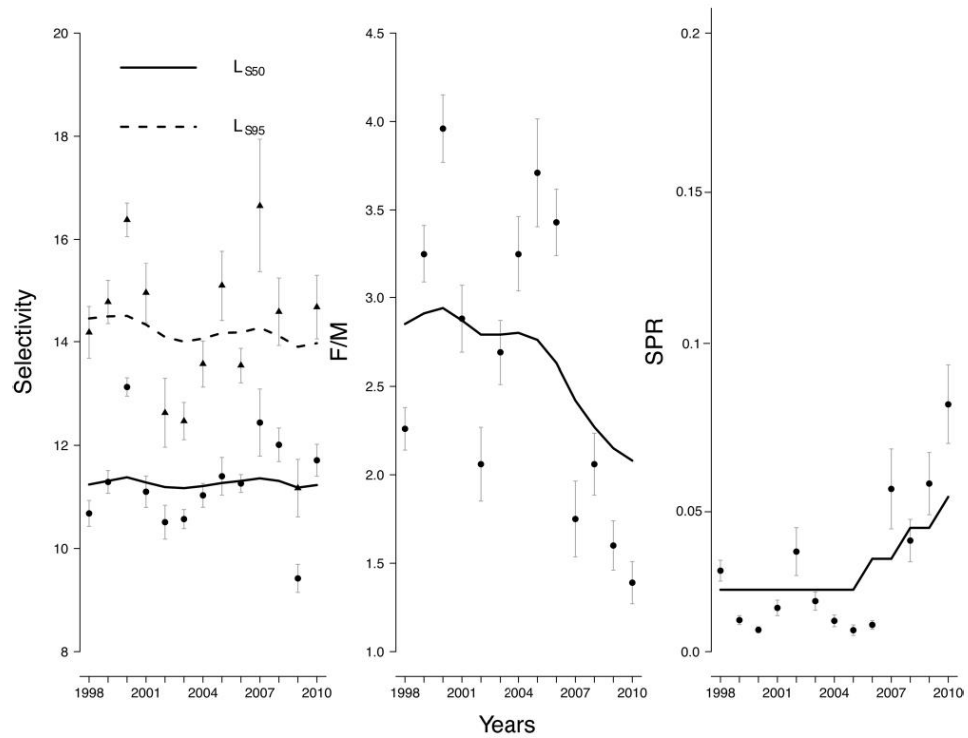


Fig. 3. LB-SPR outputs for the life-history parameters estimated for the Murcia region. Annual (individual symbol) and smoothed model (black line) estimates of selectivity,  $F/M$  and  $SPR$ . Confidence intervals (95%) are shown in vertical bars.

Poor fits for some years did not prevent the  $SPR$  estimates for the four sets of life-history parameters to be consistently below 5% from 1998 to 2006. Since 2007, the  $SPR$  tended to increase but was never higher than 8%. As for  $F/M$  estimation, the estimate of  $SPR$  did not depend on the growth hypothesis, but rather on  $M/k$ . The LB-SPR model was relatively insensitive to a  $CV$  of  $L_{\infty} < 0.15$ , but higher values

caused differences in the  $SPR$  estimate that were greater than 10% (Fig. 5).

The estimates of  $F/M$  and  $SPR$  were not very sensitive to the uncertainties in the life-history parameters, on year-to-year basis (Fig. 6a, b). The size frequency data grouped over three- or four-year periods substantially reduced the overall  $F/M$  ratio, which ranged from 1.5 to 4

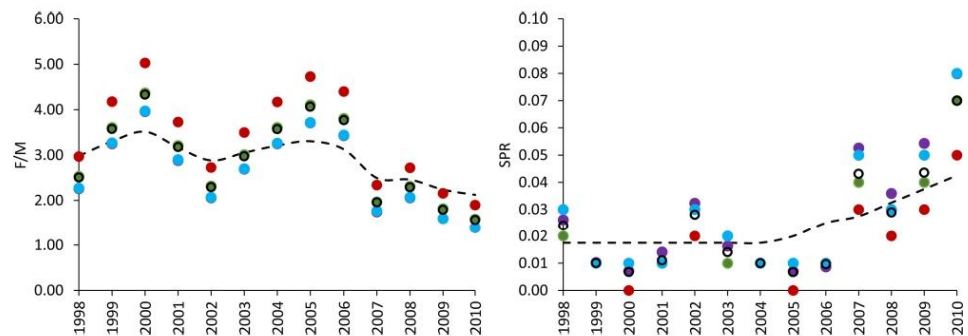


Fig. 4.  $F/M$  (left panel) and  $SPR$  (right panel) estimated using growth parameter estimated during this study (purple circles), by Aldebert and Recasens (1996) (red circles), from model 1 of Mellon-Duval et al. (2010) (green circles), and from model 3 of Mellon-Duval et al. (2010) (blue circles). Mean individual year (black empty circles) and mean smoothed model (dotted lines) estimates are shown. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

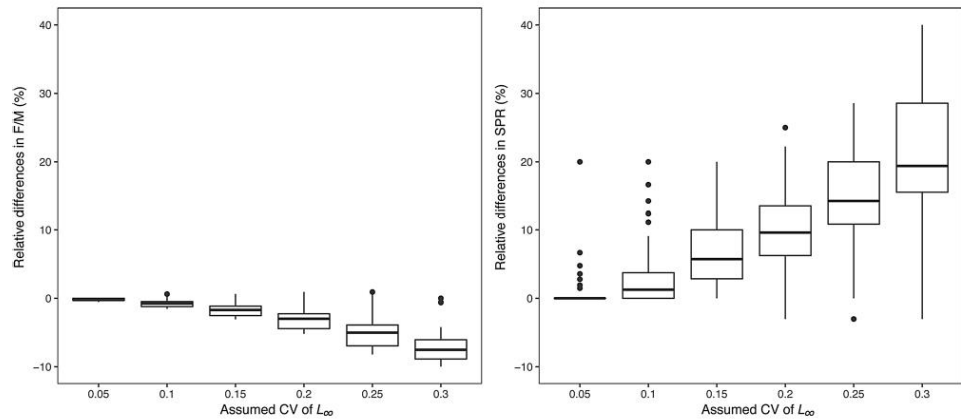


Fig. 5. Differences in estimates of  $F/M$  and  $SPR$  for various assumed values for the  $CV$  of  $L_{\infty}$  from the estimates when this  $CV$  is set to 0.001.

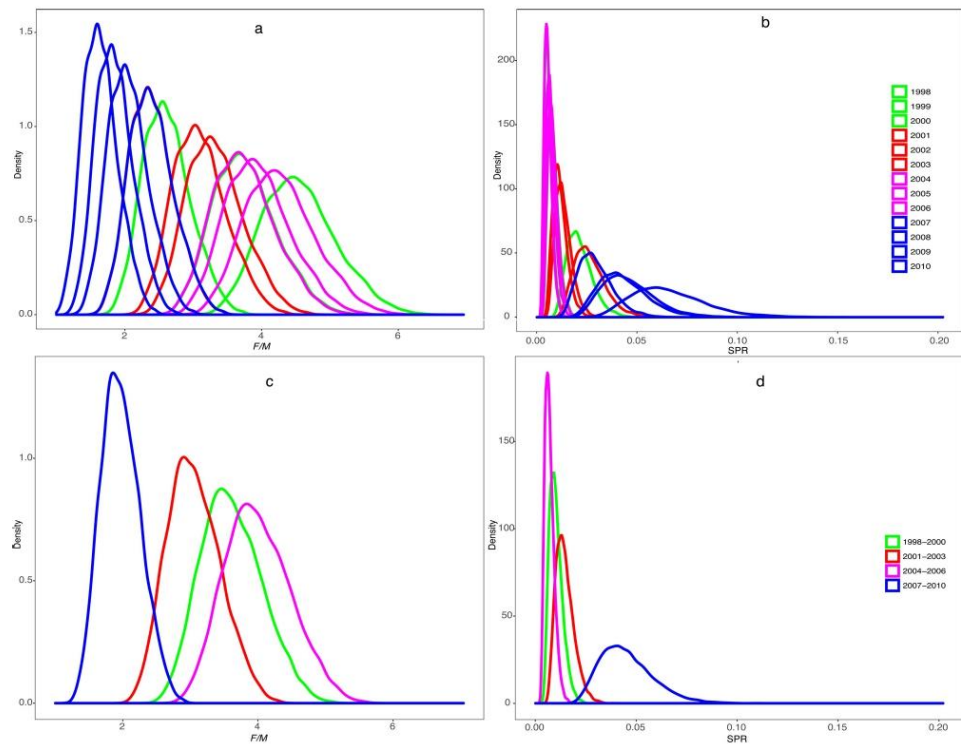


Fig. 6. Estimates of  $F/M$  (a,c) and  $SPR$  (b,d) for the European hake in the southwest Mediterranean. Results are shown by year (a,b) and when data are grouped (c,d).

(Fig. 6c). Thus, the relative fishing mortality estimated by the LB-SPR model decreased by more than 42% when data were grouped. However, this reduction in  $F/M$  did not translate into changes in  $SPR$  estimates (Fig. 6d), and comparable  $SPR$  values were observed for annual (Fig. 6a) and year-grouped (Fig. 6d) assessments.

4. Discussion

We investigated the year-to-year variability in length-based estimates of the  $SPR$  for hake, a species with moderate data availability. We explored the effect of fast- and slow-growth hypotheses and accounted

for the uncertainties of life-history parameters on annual SPR estimates. The similarity of the annual estimates of the SPR under the four life-history parameters depended strongly on how well the model fitted the size data (Fig. 2), while the SPR estimates were not sensitive to the growth hypotheses.

The fast-growth hypothesis for West Mediterranean hake (García-Rodríguez and Esteban, 2002; Mellon-Duval et al., 2010) is compatible with high relative frequencies of the youngest ages (Mellon-Duval et al., 2010). Older ages from an unfished stock would be less well-represented, corresponding to stocks exhibiting an  $M/k$  value  $> 1.5$  (Hordyk et al., 2015a,b). Aldebert et al. (1993) found that the individuals landed by bottom trawl, gillnet and longline fleets in 1988 were mostly smaller than 50 cm. Although young ages have been highly fished by the bottom trawl fishery since at least the 1950s, large individuals were already rare at that time (Bas et al., 1955). Thus, it was not unexpected that individuals near asymptotic size ( $> 90$  cm) were not recorded in our study nor in previous stock assessments (Aldebert et al., 1993; STECF, 2015).

Natural mortality of hake in the Mediterranean Sea is often estimated using PRODBIOM. Abella et al. (1997) suggest that production of hake is sustained despite high trawling pressure on juveniles, by accepting the “spawning refuge” concept for older ages. PRODBIOM produced average  $M$  estimates for the fast-growth hypothesis that were approximately twice those produced for the slow-growth hypothesis, coinciding with the rationale for choosing  $M$  for hake recommended by ICES (2014). Although PRODBIOM uses several life-history parameters ( $L_{\infty}$ ,  $k$ , estimated age and the  $a$  and  $b$  values from the length-weight relationship) to estimate natural mortality, the mean  $M$  estimates are strongly related to the  $k$  value ( $r^2 = 0.87$ ). This result implies that the concept of the fast- or slow-growth hypothesis virtually disappears when  $M/k$  instead of  $M$  and  $k$  separately is used. Thus, two growth hypotheses produced fairly similar  $M/k$  values (Table 1).

Mellon-Duval et al. (2010) questioned the reliability of the early estimates of age-structure of hake in the Mediterranean after finding evidence for fast instead of slow growth. However, Bertignac and De Pontual (2007) found that a bias in estimated ages did not affect either trend in fishing mortality nor the stock status of hake in the Bay of Biscay. On the other hand, the LB-SPR method estimates the ratio of fishing mortality to natural mortality from length-composition data, which are assumed to be representative of the exploited stock. Therefore, the SPR estimate overcomes the uncertainty associated with growth, but requires the identification of appropriate  $M/k$  and  $L50/L_{\infty}$  ratios.

$M/k > 2.3$  for the hake stock in the Murcia region corresponds to unfished stocks dominated by young classes (Hordyk et al., 2015b). These  $M/k$  values may be compatible with a “natural” stock that is mainly represented by young individuals, as can be expected in a species that has faced severe overexploitation for decades. Stock assessments for this kind of population, with low representation of adult classes, would be less sensitive to the possible bias introduced by ignoring dome-shaped selectivity (Hordyk et al., 2015c). The absence of large individuals in the size frequency would not result in large underestimates of the SPR if selectivity was not dome-shaped. However, we cannot rule out the possibility of dome-shaped selectivity, i.e., the apparent absence of large individuals from bottom trawl catches due to variation in catchability for biological reasons (Mahévas et al., 2011). Specifically, diel vertical migrations of larger individuals and/or their preferential distribution in non-trawled areas may result in dome-shaped selectivity because Spanish Mediterranean trawls are permitted to operate during daytime hours only.

As a species with moderate data availability, the asymptotic lengths of Mediterranean hake are reasonably well-known (Aldebert and Recasens, 1996; Mellon-Duval et al., 2010). The largest and lowest estimates used in this study, representing the two growth hypotheses, did not exceed 10% of the asymptotic length estimated for the Murcia region ( $L_{\infty} = 105.87$  cm). In addition, a coefficient of variation of  $L_{\infty}$

lower than 0.3 had small impacts on the LB-SPR outputs in our study, coinciding with the findings of Hordyk et al. (2015c). Thus, the probabilities of underestimating or overestimating  $F/M$  or affecting the estimate of SPR due to inaccurate or imprecise estimates of  $L_{\infty}$  were low (Hordyk et al., 2015c).

The performance of the LB-SPR method in assessing fish species from Fiji (Hordyk et al., 2015c; Prince, 2017) compares favourably with our results. In these assessments sample sizes of  $\sim 1000$  individuals were required to produce robust estimates of the SPR. Even though our sample sizes were larger than 1000, the SPR estimates from the four life-history parameter sets varied among years (from 1% to 6% approx.). The differences are likely insignificant since the SPR estimates are far ( $SPR < 10\%$ ) from the target level for a sustainability fishery in any case ( $SPR > 40\%$ , Myers et al., 1994). In the framework of fisheries management using a harvest control rule, even small changes to the current SPR could mean additional time to return to sustainable levels because more than one recovery trajectory may occur (Goodyear, 1993; Hordyk et al., 2015a).

The years when the four life-history sets produced virtually the same SPR values were derived from large samples ( $> 2000$  individuals) and corresponded to most of the best fits to the annual length distributions. These results suggest that the life-history parameters of hake that were explored in this study ( $L_{\infty} > 100$  cm,  $0.11 \text{ yr}^{-1} < k < 0.24 \text{ yr}^{-1}$  and  $0.28 \text{ yr}^{-1} < M < 0.63 \text{ yr}^{-1}$ ) and the possibilities of either fast or slow growth lead to similar LB-SPR estimates if the length structure properly represents the population.

Regardless which of the four life-history parameter sets was used, the SPR was never higher than 8% during the study period. Comparable SPR values were obtained accounting for the uncertainties of the estimates of  $L_{\infty}$ , the CV of  $L_{\infty}$ ,  $M/k$  and maturity-at-size. Nevertheless, the range of SPR estimates was quite different between the early and late years, indicating a relatively precise SPR lower than 2.5% before 2006 (except for in 1998 and 2002) and more variable and increasing SPRs since 2007.

Although an evaluation of whether hake in the West Mediterranean are out of equilibrium is beyond the scope of this study, hake recruitment in the West Mediterranean (GSA: 01, 05, 06 and 07, see STECF, 2015) from 2003 to 2014 showed moderate variability ( $\sigma_R = 0.33\text{--}0.41$ ). Hordyk et al. (2015c) found that the LB-SPR method performed relative well when  $\sigma_R < 0.6$ , producing estimates within 30% of the true value. This result matched the lower variation of  $F/M$  and the comparable values of the SPR estimates when the size compositions were grouped into three- or four-year periods. Thus, moderate changes in recruitment, which probably led to transitory population size structures, did not prevent the reliable SPR estimates for hake.

Most of the hake historically removed by the trawler fleet in the Murcia region were of sizes much less than the asymptotic length. In addition,  $L_{50}$  and  $L_{95}$  correspond to 37% and 47% of the length-at-maturity ( $L50 = 30$  cm), respectively. Sustained exploitation under this fishing regime likely explains why the SPR remained at or below 5%. The SPR increased to 8% towards the end of the study period despite the fact that the selectivity did not improve. In the absence of the knowledge of temporal changes of catchability, increased growth rates due to a favourable trophic environment (or decreased mortality) could be hypothesized.

A persistently very low SPR together with historical exploitation mostly concentrated on younger individuals raise an obvious question. What has prevented the collapse of the Western Mediterranean hake fishery? Caddy (2015) suggested that large individuals may be unavailable to the fishing gear and protected in “spawning refugia”. This would imply that both conventional (STECF, 2015) and data-limited (e.g., this study) stock assessment methods would overestimate fishing mortality and underestimate spawning biomass.

A complementary interpretation of very low and persistent SPR values should be considered for management purposes. The low SPR values provide evidence of excessive fishing pressure on the hake stock,

mainly the juveniles. This result may imply that the population size is at risk of decreasing even further because the refuge for older ages may not continue to operate to compensate for the current fishery pressure as occurred during early stages of overexploitation. Therefore, both reducing the fishing pressure on young ages and allowing the recovery of the spawning stock biomass is desirable. Thus, advice that is based on a reference point for juveniles (< age 2) as already suggested (Caddy, 2015; Colloca et al., 2013) makes sense for Mediterranean hake stocks.

## 5. Conclusions

We estimated the SPR of hake, a species with moderate data availability (appropriate knowledge of life history traits and length frequency distributions based on large samples over several years). The LB-SPR estimates were not affected by the assumed growth hypothesis since  $M/k$  leads to similar values regardless of the assumed growth parameters. However, variation in model fits and consequently estimates of  $F/M$  and SPR depend on sample size and representation of the length structure distribution of the fished stock. Uncertainty in life-history parameters (triangular distribution) led to SPR estimates that were comparable to those derived from “well-known” values (four life-history parameters sets individually used). However, fishing mortality will be overestimated, and SPR underestimated if exploitation is focused on juveniles while older individuals are partially inaccessible to fisheries. In these cases, the LB-SPR outputs should be carefully interpreted and additional reference points, related to well-recorded sizes (e.g., juveniles), should be considered.

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## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <https://doi.org/10.1016/j.fishres.2018.04.003>.

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Full length article

## Improving stock assessment and management advice for data-poor small-scale fisheries through participatory monitoring



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### ABSTRACT

Undetected but underlying biases in model parameterization strongly reduce the reliability and value of assessments of data-poor fisheries. We explore the effects of missing and misunderstood data on single-species stock assessments used to provide management advice. From 2006 to 2014, the Colombian government monitored landings of small-scale fisheries. During the same period, communities implemented a participatory monitoring program in the Central Guajira region. We found that the two data sources gave different results for the population status of the highest-valued fish, lane snapper (*Lutjanus synagris*), and the largest-landed species, white grunt (*Haemulon plumieri*). Recordings of landing points by the government monitoring program led to year-to-year underestimations and therefore misconceptions regarding population status and fishery trends. Overexploited and underexploited population statuses were seen to arise from the same fishing pressure as a result of the interplay between natural mortality and erroneous estimates of fishing mortality. The tested von Bertalanffy growth parameters affected the exploitation level, but not the population status, of the species. When data from the participatory monitoring program were incorporated, higher landings and a more severe overfishing trend emerged for both species. The management scenarios simulated using the best verified data available provided reasonable advice for recovering the lane snapper and white grunt populations. Furthermore, simulation of management measures sustained the employment and incomes of fishers. Our findings indicate that participatory monitoring should be incorporated into the assessment and management of data-poor resources.

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### 1. Introduction

Most of the world's fish stocks are either lacking sufficient information or already depleted (Costello et al., 2012; Worm and Branch, 2012). This situation is critical because a large proportion of fisheries are located in developing regions, where increased demand for food and employment coincides with a weak control of fish stocks (Salas et al., 2007). Therefore, suitable integration and interpretation of available data on fisheries and stock status are key challenges. Both the quantity and quality of data on effort, landings and catch composition are important for assessing data-poor

resources (Chen et al., 2003; Omori et al., 2016). Having fewer, but more reliable, fishery-dependent data could allow limited, but more robust, management advice to be provided for some species. Conversely, misconceptions of the real scope of available data may lead to misleading estimates of the historical and current pressures on fisheries.

The assessment of fisheries involves dealing with several sources of uncertainty (Scott et al., 2016). These uncertainties are usually interwoven with the underlying bias of the fishery model assumptions for evaluating target species in data-poor conditions (for instance, steady state; Rätz et al., 2010). Performing sensitivity tests and using the “Robin Hood” approach, which borrows data from the most reliable and related stocks, are recommended for improving model parameterization (Carruthers et al., 2014; Kokkalis et al., 2015). However, data-poor fisheries are challenged by unreported catches (Salas et al., 2007; Zeller and Pauly, 2007), leading to unpredictable estimates of trends in stock size and fishing effort (Omori et al., 2016). Consequently, misconceptions of the

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definition of fishing mortality result in highly risky results (Thorson and Prager, 2011), usually matched by a biased estimate of the life-history parameters of the studied targets from fishery-dependent data. This bias stresses the importance of “correctly” estimating natural mortality (Clark, 1999; Kenchington, 2013). Therefore, cumulative bias from multiple sources can magnify the uncertainty associated with the single-species stock assessment of data-poor fisheries (Smith et al., 2009).

The limitations of data-poor assessments appear to be sufficiently identified but poorly addressed. This occurs because stock assessment involves parameterizing a model, but the inputs cannot be verified in a data-poor context. Frequently, unverified information is the starting point for trying to improve the available data. For instance, the reconstruction of landings does not usually consider the impact of fishing effort 1) among fleets, 2) on landing points, or 3) on changes in fishing regime (Lindop et al., 2015; Wielgus et al., 2010; Zeller and Pauly, 2007). Thus, reconstruction of landings is based on several assumptions whose validation is often impossible (Chen et al., 2003).

To address these issues, we take as a case study the Amerindian Wayuu fishery located in La Guajira in the Colombian Caribbean. While it is recognized that fishery landings in the Central Guajira occur at many points along the 90-km stretch of coastline (Manjarrés, 2004), from 2006 to 2014, the government monitoring program (GOV) and subsequent management advice focused exclusively on urban areas (CCI, 2006; Puentes et al., 2014). During the same period, the Wayuu people conducted a participatory monitoring program (PAR) to record catches at most landing points, including urban and rural ones.

We explored the reliability of advice derived from single-species stock assessment for this data-poor fishery. To this end, we studied the performance of stock assessment associated with the official data provided by the GOV as a single entity and combined with the PAR data. We distinguish between fisheries with unverified data, also known as “data-random” fisheries, which can lead to misleading advice, and “data-poor” fisheries, in which the available data are optimized (Bentley, 2015).

The Wayuu fisheries are multi-species and multi-gear (Puentes et al., 2012). However, we selected the white grunt, *Haemulon plumieri*, the most important demersal species according to the landings, and the lane snapper, *Lutjanus synagris*, which fetches the highest market price among fish (CCI, 2006; Manjarrés, 2004), for this study. These two species are of great interest to Wayuu fishers, as the lane snapper covers income expectations and the white grunt meets social and food demands (J. Ramirez Pers. Comm.). We studied the effect of the GOV and PAR on the inputs of the length-based stock assessment models for both species (Leonart and Salat, 1997; Rätz et al., 2010). Here, we paid special attention to the impact of the reported landings, von Bertalanffy growth parameters (*vBGP*), and natural mortality (*M*) (Abella et al., 1997; Gislason et al., 2010; Kenchington, 2013). Stock assessments of the lane snapper and white grunt were performed using population analysis based on the interaction among gear over a limited time series. Finally, we analyzed several management scenarios aimed at achieving a sustainable fishery for both species in parallel. The harvesting sizes, fleet deployment, catch volume, food security and incomes of local communities were taken into account in the development of the management simulations.

## 2. Methods

### 2.1. Study area

La Guajira is the northernmost region of South America (Fig. 1). Its central area has a wide and shallow continental shelf influenced

by upwelling (Andrade and Barton, 2005; Sarmiento-Devia et al., 2013). These conditions promote high fishing activity (Manjarrés, 2004). Artisanal fisheries are dominant in the area, whereas trawl and larger long-line fleets are scarce (CCI, 2010; Zuñiga et al., 2004).

### 2.2. Data collection

The GOV database provided total landings by day from 2006 to 2011, but not the number of vessels. From November 2012 to July 2014, daily landings and the number of vessels were recorded. Throughout the study period, the catches of all vessels that arrived at five urban landing points in the Central Guajira were recorded (CCI, 2006). Data collection for the target species included the length frequency and macroscopic definition of maturity (CCI, 2006). For lane snapper, this information was only available in 2013. Little biological data for white grunt were collected because this species is not prioritized in the official fisheries advice (Barreto and Borda, 2008; Puentes et al., 2014).

A participatory monitoring program implemented by a local non-governmental organization (Fundación Ecosfera) and the Wayuu people was conducted during the same period. Through the participation of 19 communities, information from 85,347 daily trips (2006–2014) was recorded, based on over 18 rural and 3 urban landing points. The trained members of each community randomly collected landing data on four days a week. The total landings by species from the PAR were estimated using a daily raising factor – the number of sampled to unsampled active vessels. Additionally, the PAR collected length frequencies and maturity information for white grunt and lane snapper in 2010.

### 2.3. Data treatment

#### 2.3.1. Landings reconstruction

Interpolation from reliable data was used to fill the gaps in landings for which the period and landing point were missing, following the methodology established by the Sea Around Us Project (Zeller and Pauly, 2007). Three landing points in urban areas, sampled at the same time by both the government and participatory monitoring programs, were used to determine the degree to which landings in urban areas were underestimated. The total landed catch and selectivity pattern by gear of both species were calculated separately for the GOV and the PAR. Because daily landings by each vessel were not available for the GOV from 2006 to October 2012, the catch per unit effort (*CPUE*) was annually calculated from total landings and active vessels by fleet. Additionally, by combining the available length frequencies and landings by fleet, we were able to estimate the average size of the fish caught per year.

#### 2.3.2. Life-history parameters

The *vBGP* were estimated using a non-linear model fit to the modal progression data with the Bhattacharya method, the selected asymptotic length ( $L_{\infty}$ ) and the growth constant rate (*k*) according to the FISAT II routines (Gayanilo et al., 2005; Pinheiro et al., 2015). The *vBGP* of lane snapper were obtained from the encircling gillnets ( $n = 1271$ ) of the GOV in 2013. Using data from the bottom gillnets ( $n = 541$ ), long-lines ( $n = 1055$ ) and encircling gillnets ( $n = 99$ ) of the PAR, an additional set of *vBGP* were calculated. In the case of the white grunt, length frequencies were only available from the PAR study. Here, bottom gillnets ( $n = 977$ ), long-lines ( $n = 181$ ) and encircling gillnets ( $n = 15$ ) provided the data to estimate the *vBGP*.

The *vBGP* calculated from fishery-dependent data on a population subjected to strong exploitation are probably biased (Then et al., 2015). We therefore explored several growth parameters for lane snapper derived from the literature (Acosta and Appeldoorn, 1992; Barreto and Borda, 2008; Frédoou et al., 2009; Gómez et al., 2001; Johnson et al., 1995; Leite et al., 2005; Luckhurst et al., 2000;

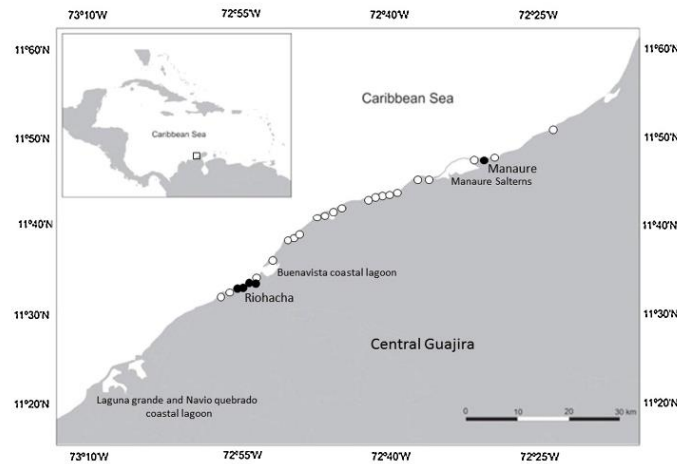


Fig. 1. Map of central Guajira, showing its position in the Caribbean Sea. Urban areas where the national fishery agency records landings are shown by filled circles, and rural areas where the Wayuu people performed the participatory monitoring are shown by empty circles.

Manickchand-Dass, 1987; Manjarrés, 2004; Puentes et al., 2014). Only the  $vBGP$  obtained in Trinidad and Tobago by Manickchand-Dass (1987) for females met the conditions of 1) being based on otolith readings, 2) showing an asymptotic length higher than the maximum length found in the Central Guajira, 3) including small and large sizes and showing environmental conditions similar to our study area, and 4) being insensitive to changes in the value of terminal fishing mortality (explained below). However, both the  $L_{\infty} = 60$  cm and  $k = 0.20$  yr<sup>-1</sup> obtained from this study were similar to our own ( $L_{\infty} = 64.94$  cm,  $k = 0.19$  yr<sup>-1</sup>). To explore a wider range of plausible states of nature, the sensitivity analysis further considered  $L_{\infty} = 63.75$  cm and  $k = 0.27$  yr<sup>-1</sup>, which were previously used in Colombian assessments (see Puentes et al., 2014). Using the above criteria for lane snapper, a second set of  $vBGP$  for white grunt was selected from the available literature (Araújo and Martins, 2007; Dominguez-Viveros, 1996; Murie and Parkyn, 2005; Potts and Manooch, 2001; Valle and García-Arteaga Claro, 1997). None of the  $vBGP$  met all the criteria. Therefore,  $L_{\infty} = 59.10$  cm and  $k = 0.08$  yr<sup>-1</sup> (Potts and Manooch, 2001) were explored, though they belong to a population that inhabits colder water.

Given its high uncertainty (Kenchington, 2013; Scott et al., 2016; Then et al., 2015),  $M$  for both species was calculated using several indirect estimators. Generic  $M$  estimators based on life-history correlates, such as Pauly's estimator and Ralston's estimator (focusing on lutjanids) were explored (Kenchington, 2013). Hoenig's estimator based on maximum estimated ages ( $T_{max}$ ) was used (Newman, 2002; Newman et al., 2000), as were estimators of  $M$ -at-age (Abella et al., 1997) and  $M$ -at-size (Gislason et al., 2010).

$M$  estimators based on  $T_{max}$  were applied based on the oldest ages from the literature. The ages were considered valid when they came from populations that were either lightly exploited or not overexploited. The oldest selected age for lane snapper was 19 years (Luckhurst et al., 2000). Additionally, a natural mortality of 0.13 yr<sup>-1</sup>, estimated by the catch-curve method for unfished populations of other lutjanids, was included in the analyses (Newman, 2002; Newman et al., 2000). Considering that large differences in the oldest age of white grunt have been found (see Araújo and Martins, 2007; Murie and Parkyn, 2005), we considered maximum ages of 18 and 28 years for the species.

The data used to calculate female maturity for lane snapper ( $n = 504$ ) and white grunt ( $n = 660$ ) came from the PAR data set. Individuals were considered mature as of stage II (developing) by macroscopic observation because lane snapper and white grunt can begin to spawn when they have reached the developing stage (Freitas et al., 2014; ICES, 2008; Shinozaki-Mendes et al., 2013). Different analytical procedures for obtaining the maturity-at-length curve were explored (ICES, 2008). The logistic function is usually used for fitting the proportion of mature fish by size (or age) and obtaining parameters, particularly  $L_{50}$ . The cumulative normal distribution fits, as well as the logistic function, has the advantage of being more theoretically justifiable, as maturation occurs following a probability law. However, both models are theoretically problematic, as negative values of the independent variable, which are not defined, yield positive values of maturation, which is nonsensical. Thus, it is more justified to fit a lognormal cumulative curve that meets all the theoretical requirements and explains the maturation process equally well or better. The cumulative lognormal gave the best fit to the data with the following maturity-at length parameters for the lane snapper ( $L_{25} = 18.94$  cm,  $L_{50} = 22.77$  cm,  $L_{75} = 29.61$  cm) and the white grunt ( $L_{25} = 15.39$  cm,  $L_{50} = 16.33$  cm,  $L_{75} = 19.21$  cm). Weight-at-length relationships were calculated from the PAR data using  $W = aL^b$  (where  $a = 0.0173$  and  $b = 2.9168$  and  $a = 0.008$  and  $b = 3.1715$  for the lane snapper and white grunt, respectively). Given limited data for calculating maturity-at-length, the sensitivity tests performed in this study do not account for the uncertainty associated with maturity ogive calculation.

### 2.3.3. Sensitivity tests and population dynamics

We assumed that the size structure of species caught with the three types of fishing gear did not change during the study period because length-frequency data were only available for the bottom gillnet and long-line in 2010 and for the encircling gillnet in 2013. The stock assessment was performed using the VIT software, which was specially developed to conduct stock assessment when catch-at-length data are limited in duration (Leonart and Salat, 1997). Virtual population analysis (VPA), yield per recruit ( $Y/R$ ) and management simulations were performed using catch data structured by age (Aldebert and Recasens, 1996; Rätz et al.,

2010). Furthermore, length data, maturity-at-length and mortality-at-length were converted to age data using a slicing method based on the von Bertalanffy growth equation. The biological process could thus be propagated through age-based data.

Since the age groups of a given year are used to obtain a single cohort from pseudo-cohort assumptions, one VPA was performed for each individual year (Leonart and Salat, 1997; Rätz et al., 2010). The terminal fishing mortality ( $F_t$ ) is required to start the VPA for each individual year, so we performed a sensitivity analysis of this parameter (0.05, 0.1, 0.2, 0.3, 0.4, 0.5, 1.0 and 1.5 yr<sup>-1</sup>) on the VPA results, keeping the same  $vBGP$  and  $M$ . Those  $vBGP$  that led to the same fishing mortality trend among ages when  $F_t$  changed were chosen, and the others were rejected. Terminal fishing mortalities from 0.05 to 0.5 yr<sup>-1</sup> led to similar stock results, so an intermediate value of  $F_t = 0.2$  yr<sup>-1</sup> was selected for both species. The plus group ranged from two to four for lane snapper and from five to six for white grunt, depending on the  $vBGP$  used.

According to the selected  $vBGP$  for each species and natural mortality estimated by different indirect estimators, as mentioned above, we conducted sensitivity analyses to explore the parameters' impact on assessment outputs. For these analyses, we used the data from 2006 as the first year with available data of landings and length structure by fleet. From the three sets of  $vBGP$  and six estimates of natural mortality, the sensitivity analyses of lane snapper included 18 VPA and Y/R. In the case of the white grunt, we used three  $vBGP$  and five different estimates of natural mortality, producing 15 VPA and Y/R. During the sensitivity analyses, only the  $vBGP$  and  $M$  were modified, keeping the other inputs fixed. The sensitivity results were plotted using an adapted Kobe plot (Nishida et al., 2014). Virgin biomass estimators using VIT are uncertain and not very reliable (Rätz et al., 2010). However, these authors found that the overall trends of fishing mortality and stock size by VIT are well captured in comparison to a traditional VPA. To make the stock assessment comparable among years, we selected, as a relative measure of exploitation, the ratio between current fishing mortality and that which produces a maximum sustainable yield ( $F/F_{msy}$ ). For the abovementioned reason, we consider the  $F_{max}$  and its yield as proxies for  $F_{msy}$  and  $MSY$ . Additionally, we considered two relative reference points according to the maximum (virgin) spawning stock biomass estimated in each year ( $SSB/SSB_{max}$ ):  $SSB_{40\%}$ , as a precautionary reference point when stock recruitment relationships were unknown (Myers et al., 1994), and  $SSB_{20\%}$ , defined as the fishing level that reduces spawning per recruit to a threshold biomass level of 20% of the unfished stock (Rosenberg, 1993).

### 2.3.4. Management strategy simulations

The initial age-structure at year 0 to start the management simulations was defined according to the average of the last two years of the reconstructed data (2013 and 2014). We took into account control of the fleet effort (which, in the model, acts as an  $F$  multiplier,  $\phi$ ) in terms of vessel numbers, incomes, and food security for the Central Guajira communities. Considering it mandatory to reach a relative  $SSB_{20\%}$  within 8 years, the current  $F/F_{msy}$  was adjusted by changing the fishing multiplier by gear. We explored several scenarios for managing the lane snapper and white grunt (Table 1). A change in fishing effort by gear could be not required ( $\phi = 1$ ), increased ( $\phi > 1$ ), reduced ( $\phi < 1$ ), or gear eliminated ( $\phi = 0$ ). The management simulations also studied the effect of eliminating the effort of two fleets and allowing the optimization of the third fleet effort to achieve  $MSY$ . For instance, the "people first" scenario prioritizes gillnets because it is the gear most used by rural fishers and it catches medium-sized fish. In this case, the efforts of long-line and encircling gillnet fleets are reduced to zero, whereas the fishing effort of gillnets is increased as much as possible.

Simulation under different life-history and stock-recruitment models extended the sensitivity analyses. Two lane snapper  $vBGP$

sets were selected:  $L_\infty = 64.94$  cm,  $k = 0.19$  yr<sup>-1</sup>,  $t_0 = -1.58$  yr and  $L_\infty = 63.75$  cm,  $k = 0.27$  yr<sup>-1</sup>,  $t_0 = -0.66$  yr. The first were calculated from the widest range of sizes available from the PAR and the second were those used by the government assessments. However, with white grunt, only a single  $vBGP$  set from the Central Guajira was available ( $L_\infty = 41.65$  cm,  $k = 0.26$  yr<sup>-1</sup>,  $t_0 = -1.43$  yr), but a wider range of simulations was conducted based on the two calculated natural mortality rates (0.15 yr<sup>-1</sup> and 0.24 yr<sup>-1</sup>). The Beverton and Holt model (B&H) (Beverton and Holt, 1959) was used to simulate recruitment ( $R = 1/(\alpha + SSB/\beta)$ ). This model was used to simulate a constant recruitment ( $\alpha = 1, \beta = 0$ ) or a quasi-linear one ( $\alpha = 0.5, \beta = 0.5 \cdot SSB$ ). The constant recruitment was initially used because recruitments and reliable  $SSB$  by year were unknown. Additionally, a quasi-linear model was used by assuming that the management measures could be adequately implemented by stakeholders from year one and that pre-recruitment phases of target species would not be affected by the recruitment phase. The number of stochastic iterations of the lognormally distributed stock-recruitment model for each run was set at 1000 with a mean of 1 and a variance of 0.1. The yield per recruit was represented by the mean with 95% confidence intervals.

## 3. Results

### 3.1. Catch reconstruction

From 2006 to 2014, the total reconstructed catch of lane snapper was estimated at 410 tons and the total catch of white grunt at 2,017 tons. The GOV data underestimated landings. The reconstructed landings of lane snapper and white grunt ranged from 3.25 to 8.52 times and 1.38 to 3.70 times greater than the official GOV landings (Fig. 2). Most landings of lane snapper occurred in rural areas unmonitored by the GOV. In the case of white grunt, the official and reconstructed catches showed similar trends from 2006 to 2009. As of 2010, higher landings in rural areas led to the greater underestimations of government statistics (Fig. 2b). The GOV data suggested that the encircling gillnet fleet took the largest proportion of lane snapper (mean, 83%) and white grunt (mean, 99%). The reconstructed landings of lane snapper showed that the long-line fleet took the bulk of the catch during the first few years but that the catches have shifted to being dominated by encircling and bottom gillnets since 2011 (Fig. 3a and b). The reconstructed landings of white grunt increased the participation of bottom gillnets to 20% (Fig. 3c and d). On the other hand, the reduction of the average size of lane snapper remained hidden in the official GOV landings, and the reconstructed landings show the size to have decreased during the study period (Fig. 3). The average size of white grunt among the years remained stable in both monitoring programs.

### 3.2. Life-history parameters

The catches of lane snapper from the bottom gillnet and long-line fleets (from PAR) were mainly composed of sizes ranging from 14 to 40 and from 20 to 50 cm, respectively (see supplementary material, table S1). The encircling gillnet fleet (from GOV) mostly included sizes ranging from 19 to 41 cm (see supplementary material, table S2). Consequently, the  $vBGP$  calculated from the long-lines and bottom gillnets ( $L_\infty = 64.94$  cm,  $k = 0.19$  yr<sup>-1</sup>,  $t_0 = -1.58$  yr) produced a higher  $k$  than the  $vBGP$  calculated from the encircling gillnets ( $L_\infty = 62.5$  cm,  $k = 0.15$  yr<sup>-1</sup>,  $t_0 = -3.19$  yr). The length-based analysis for the white grunt population mainly included data from bottom gillnets (see supplementary material, table S3). Thus, the white grunt ranged from 15 to 36 cm, with a high  $k$  value ( $L_\infty = 41.65$  cm,  $k = 0.27$  yr<sup>-1</sup>,  $t_0 = -1.43$  yr).

**Table 1**

Management scenarios of lane snapper and white grunt fisheries according to fish size, deployed fleet and food security and incomes of fishing communities. Management scenarios are separated by species given that the fleets act differently on sizes and total catches. Management scenarios were incorporated into simulations by controlling the caught sizes by gear (Fish size column) and fleets in the fishery (Deployed fleet column). The third column (Food security and profits of fishing communities) shows the effect caused by controlling for fish size and fleet, but it is not directly incorporated into the simulation model.

	<i>Lutjanus synagris</i> Management scenarios	Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher and high-valued catches and ensuring food security in the rural communities (long-line and bottom gillnet)
2	The “best” catches	Removing gear that fishes the highest proportion of medium sizes (bottom gillnet)	Eliminating the fleet formed by the most vessels from rural communities (bottom gillnet)	Optimizing both highly valued sizes (long-line) and the highest catches (encircling gillnet), even if food security in rural areas could be affected (bottom gillnet)
3	Only harvesting medium sizes	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and small sizes (encircling gillnet)	Keeping the fleet formed by the most vessels from rural communities (bottom gillnet)	Encouraging both the cheapest fishing trips and food security, especially in rural areas
4	The volume does matter	Reducing as much as possible catches with a high proportion of both large sizes (long-line) and medium sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
5	Targeting the highest-valued sizes	Reducing as much as possible catches with a high proportion of both small sizes (encircling gillnet) and medium sizes (bottom gillnet)	Keeping the fleet that underwent the greatest fishing reduction during the study period	Promoting the highest-valued as well as the lowest catches. Food security of rural communities is poorly considered
6	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)
	<i>Haemulon plumierii</i> Management scenarios	Fish size	Deployed fleet	Food security and profits of fishing communities
1	People first	Removing gear that fishes the highest proportion of small sizes (encircling gillnet)	Eliminating the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher catches and ensuring food security in the rural communities (bottom gillnet)
2	The volume does matter	Removing gear that fishes the highest proportion of medium and large sizes (bottom gillnet)	Keeping the fleet that exhibits the highest harvesting power (encircling gillnet)	Promoting higher landings in urban areas and not giving priority to profits and food security of rural communities
3	Lower fishing	Homogeneous reduction of catches of all sizes (all gears)	Homogeneous reduction of all fleets (all gears)	Homogeneous reduction of food security and profits of all fleets (all gears)

Natural mortality estimators based on life-history correlations tended to produce higher natural mortality rates in both species (Pauly's  $M=0.41-0.67\text{ yr}^{-1}$  and Ralston's  $M=0.30-0.42\text{ yr}^{-1}$ ). Similar results were produced by Gislason's *M-at-size* estimator (lane snapper average size,  $M=1.21-0.62\text{ yr}^{-1}$ ; white grunt average size,  $M=1.97-0.31\text{ yr}^{-1}$ ; see supplementary material, table S4). Additionally, the estimates from *M* estimators varied substantially depending on the  $L_{\infty}$  and  $k$  values used. Conversely, Hoenig's estimator based on  $T_{\max}$  produced lower or moderate natural mortality rates (lane snapper,  $M=0.23\text{ yr}^{-1}$  for  $t_{\max}=19\text{ yr}$ ; white grunt,  $M=0.15\text{ yr}^{-1}$  for  $t_{\max}=8\text{ yr}$  and  $M=0.28\text{ yr}^{-1}$  for  $t_{\max}=18\text{ yr}$ ). Abella's *M-at-age* estimator, which is also based on life-history correlations, produced lower mortality rates. The average of the *M* values for the first six ages ranged from 0.10 to 0.18  $\text{yr}^{-1}$  for lane snapper and from 0.12 to 0.20  $\text{yr}^{-1}$  for white grunt (see supplementary material table S5).

### 3.3. Sensitivity tests

The population status of both species ranged from highly over-exploited to underexploited (Fig. 4). This was mainly due to the natural mortality assumed in each stock assessment. The natural mortality estimated from Pauly's, Ralston's and Gislason's estimators yielded no overfished stocks ( $SSB/SSB_{\max}>40\%$ ) and overfished stocks ( $SSB/SSB_{\max}<20\%$ ), but overfishing was not occurring ( $F/F_{\text{msy}}<1$ ). Conversely, stock assessments based on the

*M* from Hoenig's and Abella's estimators indicated overexploitation. The exploitation levels of lane snapper and white grunt were also largely influenced by changes of *vBCP*. Abella's estimator showed the largest changes in  $F/F_{\text{msy}}$ . Gislason's, Pauly's and Ralston's estimators led to a large variation in  $SSB/SSB_{\max}$ . Hoenig's estimator, which is based on the maximum estimated age, led to the lowest variation in exploitation level (Fig. 4).

### 3.4. Population status

We chose three and two sets of *vBCP* to describe the growth of the lane snapper and white grunt, respectively. In the case of the lane snapper, the landings reduction and increased fishing mortality of small-sized fish during the study period suggested overfishing. Additionally, the high values of natural mortality contravene the low to moderate estimates of *M* obtained from unexploited snapper populations using catch-curve methods (Newman, 2002; Newman et al., 2000). For lane snapper, Frédou et al. (2009) suggested low *M*. In performing a stock assessment of the lane snapper, we selected natural mortality rates of 0.13 and 0.24  $\text{yr}^{-1}$  from the Newman et al. (2000) and Hoenig's estimators. Further complications emerge for selecting *M* for the white grunt, as larger gaps in its estimation remain when indirect estimators are used. Therefore, we selected the natural mortality rates of 0.15 and 0.24  $\text{yr}^{-1}$  by focusing on methods that use the maximum estimated

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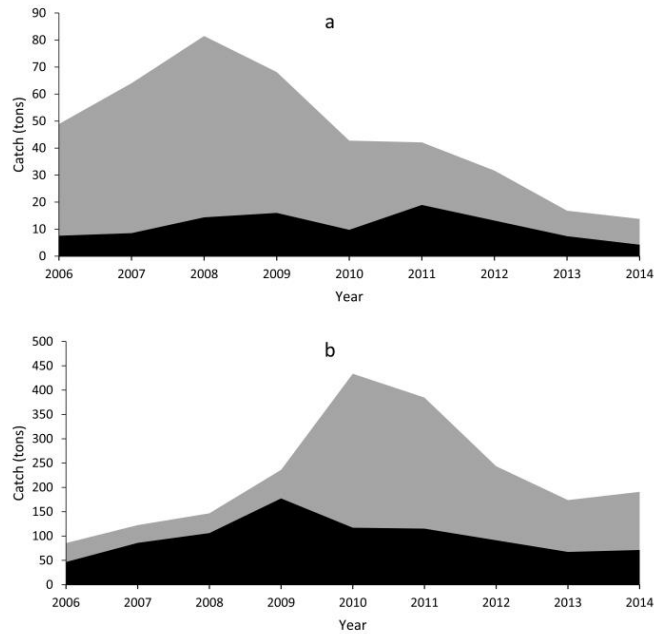


Fig. 2. Catch of lane snapper (a) and white grunt (b) reported by the national fishery agency (black) and total reconstructed catch (gray).

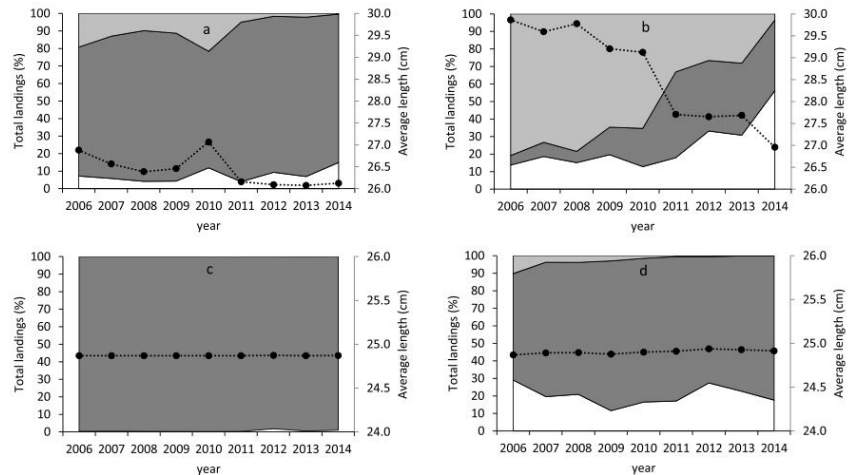


Fig. 3. Percentage of long-line (light grey), bottom gillnet (white) and encircling gillnet (dark grey) caught fish in the catches of lane snapper (a and b) and white grunt (c and d) from 2006 to 2014 (left y axis). Average size of caught individuals, on the right y axis (dotted line). Government monitoring program (a and c) and combined government and participatory data (b and d).

age, as suggested by [Then et al. \(2015\)](#), and avoided using  $M$  values higher than  $0.35 \text{ yr}^{-1}$ , as recommended by [Potts \(2000\)](#).

The lane snapper was estimated to be more overexploited when the assessment was based on landings from the GOV ([Fig. 5a](#)). Underestimation of long-line catches in urban areas meant that

large fish sizes were unreported. Furthermore, the stock assessment performed using the GOV data led to less change between earlier and later years than that obtained when the PAR data were also included in the assessment. This means that a worsening in the overexploitation status of the lane snapper was smoothed

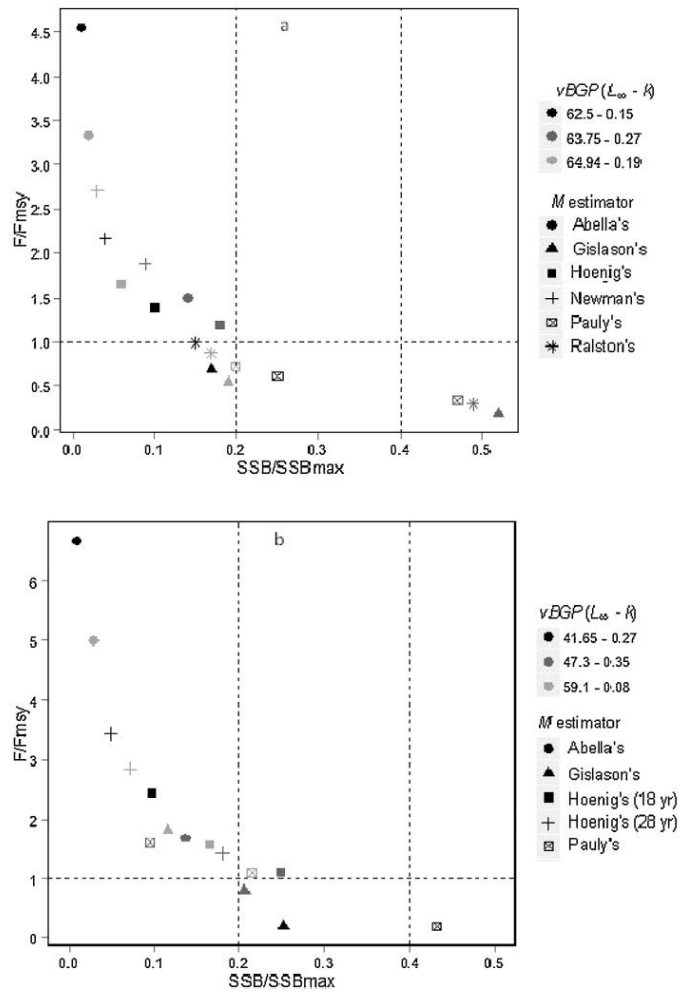


Fig. 4. Modified Kobe plot for three pairs of  $vBGP$  (symbols sharing the same color) and various natural mortality estimators (different symbols). Lane snapper (a) and white grunt (b) in the Central Guajira.

because the GOV did not perceive the increase in fishing mortality of smaller-sized fish.

The status of the white grunt population was insensitive to whether government and unified government-participatory monitoring data were analyzed (Fig. 5b). The encircling gillnet fleet showed a large participation in urban and rural landing points, reporting between 61% and 83% of the total reconstructed landings. Additionally, the encircling gillnet fleet caught fish sizes similar to the those for the bottom gillnet fleet, implying that between 89% and 99% of landings involved a similar length structure. These conditions contributed to an unchanged population status between the GOV and reconstructed data. Therefore, the total biomass harvested

by the fleet should be a key aspect to consider in the management of the white grunt population.

### 3.5. Management simulations

The lane snapper and white grunt populations are estimated to have been overexploited since 2006. The dominance of the encircling gillnet fleet in lane snapper and white grunt catches, which mainly removed small-sized fish, placed the relative spawning stock biomass ( $SSB/SSB_{max}$ ) of both species below 20%. The trends suggest higher overfishing between the earlier and later years studied (Fig. 5). The simulation outputs covered the selected  $vBGP$  and  $M$ , as well as the two different recruitment models (Constant and

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**Table 2**

Simulations of management scenarios, based on  $L_{\infty} = 64.94$  cm,  $K = 0.19$  yr<sup>-1</sup>,  $t_0 = -1.58$ ,  $M = 0.23$  yr<sup>-1</sup> (a) and  $L_{\infty} = 63.75$ ,  $K = 0.27$  yr<sup>-1</sup>,  $t_0 = -0.66$  yr,  $M = 0.13$  yr<sup>-1</sup> (b) to lane snapper and  $L_{\infty} = 41.65$  cm,  $K = 0.27$  yr<sup>-1</sup>,  $t_0 = -1.43$  yr,  $M = 0.15$  yr<sup>-1</sup> (c) and  $L_{\infty} = 41.65$  cm,  $K = 0.27$  yr<sup>-1</sup>,  $t_0 = -1.43$  yr,  $M = 0.24$  yr<sup>-1</sup> (d) for white grunt populations. Each scenario produced four results according to vBGP-M and either the constant or Beverton and Holt (B&H) recruitment models. Simulation cases in which the prioritized fleet could maintain a fishing multiplier ( $\phi$ ) above 1 were maximized, and other gears were eliminated (optimization column).

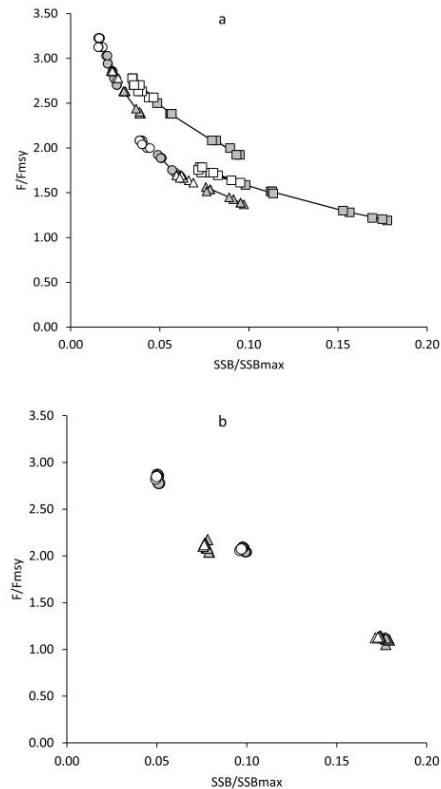
Species	Scenario	vBGP and M	Recruit model	Management measures ( $\phi$ by gear)			
				Bottom gillnet	Encircling gillnet	Long-line	Optimization of $\phi$
<i>L. synagris</i> (Lane snapper)	People first	a	Constant	0.82	0.00	0.82	N/A
			B&H	1.00	0.15	1.00	1.1
		b	Constant	0.90	0.00	0.90	N/A
			B&H	1.00	0.27	1.00	1.2
	The "best" catches	a	Constant	0.00	0.92	0.92	N/A
			B&H	0.25	1.00	1.00	1.2
		b	Constant	0.00	0.90	0.90	N/A
			B&H	0.28	1.00	1.00	1.2
	Only harvesting medium sizes	a	Constant	1.00	0.12	0.12	1.15
			B&H	1.00	0.40	0.40	1.5
		b	Constant	1.00	0.16	0.16	1.2
			B&H	1.00	0.45	0.45	1.62
	The volume does matter	a	Constant	0.21	1.00	0.21	1.35
			B&H	0.45	1.00	0.45	1.75
		b	Constant	0.20	1.00	0.20	1.27
			B&H	0.47	1.00	0.47	1.65
	Targeting the highest-valued sizes	a	Constant	0.40	0.40	1.00	2.9
			B&H	0.60	0.60	1.00	3.9
	b	Constant	0.42	0.42	1.00	3.35	
		B&H	0.63	0.63	1.00	4.7	
Lower fishing	a	Constant	0.51	0.51	0.51	N/A	
		B&H	0.65	0.65	0.65	N/A	
	b	Constant	0.52	0.52	0.52	N/A	
		B&H	0.69	0.69	0.69	N/A	
<i>H. plumierii</i> (white grunt)	People first	c	Constant	1.00	0.06	1.00	1.19
			B&H	1.00	0.22	1.00	1.7
		d	Constant	1.00	0.29	1.00	1.9
			B&H	1.00	0.46	1.00	2.6
	The volume does matter	c	Constant	0.00	0.4	1.00	N/A
			B&H	0.00	0.55	1.00	N/A
		d	Constant	0.00	0.60	1.00	N/A
			B&H	0.00	0.76	1.00	N/A
	Only harvesting medium sizes	c	Constant	0.30	0.30	1.00	N/A
			B&H	0.40	0.40	1.00	N/A
		d	Constant	0.46	0.46	1.00	N/A
			B&H	0.59	0.59	1.00	N/A

B&H). The B&H recruitment model allowed less reduction of  $\phi$  than the constant model for both species, regardless of which set of vBGP was used. The management scenarios "people first" ( $1 \geq \phi > 0.82$ ) and "only harvesting of medium sizes" ( $\phi = 1$ ) led to the lowest reduction of the bottom gillnet fleet effort (Table 2). From a social point of view, this is desirable because  $Y/R$  increased over initial simulation levels (see scenarios 1 and 3, Fig. 6). Furthermore, many fishers who inhabit rural areas would be able to secure income and food (Table 1). Conversely, if priority is given to obtaining the highest of either income or catch, the encircling gillnet fleet should be maintained (Table 1). This would, however, lead to a reduction in the effort of most bottom gillnet vessels ("the best catches";  $0.28 \geq \phi \geq 0$ ) and long-line vessels ("the volume does matter";  $0.47 \geq \phi \geq 0.2$ ). This means that rural communities would be seriously affected (Tables 1 and 2), and the recovery of the lane snapper population would be both lower and slower than in previous scenarios (Fig. 6). Focusing the fishing effort on the highly valued large fish sizes increases long-line yields. This would require an important reduction in the effort of the bottom and encircling gillnet fleets ( $0.63 \geq \phi \geq 0.40$ ), which would be socially unsustainable.

The white grunt fishery has a marginal long-line harvest (lower than 5% of total catch), so it was not necessary to consider long-lines

in the management scenarios. The "people first" scenario, which prioritized the bottom gillnet fleet, also allowed some participation of the encircling gillnet fleet ( $0.46 \geq \phi \geq 0.06$ ). This would prioritize the protection of employment and food security for rural and urban fishers (scenarios 1 and 2; Fig. 7). Conversely, prioritizing the largest catches demands a reduction in the effort of the encircling gillnet fleet ( $0.76 \geq \phi \geq 0.4$ ) and the total elimination of the bottom gillnet fleet ( $\phi = 0$ ), a total reduction of 66.3% to 86.5% of the operative fleet effort (Table 2). Finally, a homogeneous reduction of all fleet efforts for both species would not offer higher yields but would result in greater negative impacts on fishery incomes and food security (Figs. 6 and 7).

When scenarios that favored the traditional bottom gillnet fleet ("people first" and "just harvesting of medium sizes") were optimized, the fishing effort on lane snapper ( $1.62 \geq \phi \geq 1.1$ ) and white grunt ( $2.6 \geq \phi \geq 1.19$ ) could be increased. Optimization of a more dominant fleet, the encircling gillnets ("the volume does matter"), allowed the fishing effort on lane snapper to be increased ( $1.75 \geq \phi \geq 1.27$ ). However, this optimization would require harvesting of white grunt to be reduced to achieve  $SSB_{20\%}$  ( $\phi = 0$ ) (Table 2), suggesting that joint management of both species



**Fig. 5.** Plot of the relative spawning stock biomass ( $SSB/SSB_{max}$ ) versus the  $F$  ratio ( $F/F_{msy}$ ). Analysis performed using government landings (empty symbols) and reconstructed landings (gray symbols). Lane snapper (a) when  $L_{\infty} = 64.94$  cm,  $k = 0.19 \text{ yr}^{-1}$ ,  $M = 0.23 \text{ yr}^{-1}$  (lowermost circles) and  $M = 0.13 \text{ yr}^{-1}$  (upper circles);  $L_{\infty} = 63.75$  cm,  $k = 0.27 \text{ yr}^{-1}$ ,  $M = 0.23 \text{ yr}^{-1}$  (lowermost squares) and  $M = 0.13 \text{ yr}^{-1}$  (upper squares); and  $L_{\infty} = 63.75$  cm,  $k = 0.27 \text{ yr}^{-1}$ ,  $M = 0.23 \text{ yr}^{-1}$  (lowermost triangles) and  $M = 0.13 \text{ yr}^{-1}$  (upper triangles). White grunt (b) when  $L_{\infty} = 41.65$  cm,  $k = 0.26 \text{ yr}^{-1}$ ,  $M = 0.24 \text{ yr}^{-1}$  (lowermost triangles) and  $M = 0.15 \text{ yr}^{-1}$  (upper triangles); and  $L_{\infty} = 59.01$  cm,  $k = 0.08 \text{ yr}^{-1}$ ,  $M = 0.24 \text{ yr}^{-1}$  (lowermost triangles) and  $M = 0.15 \text{ yr}^{-1}$  (upper triangles).

depends on the fishing effort of the encircling gillnets on the white grunt population.

#### 4. Discussion

##### 4.1. Addressing reliable biomass removal by fishery

The Wayuu fishery in the Central Guajira of Colombia brought to light a large underestimation obtained from the official catch statistics. The reconstructed landings tended to decrease, but this behavior was not detected by the GOV. This is because the GOV only considered major urban landing points, whereas fishing vessels are also landing on a wide rural shoreline, as occurs in many developing countries (Salas et al., 2007). Inter-annual changes in the landing areas according to fleet also affected the reported catches. The long-line fleet (which decreased during the study period) and the bottom gillnet fleet mainly landed in rural areas. The encircling gillnet fleet

landed in urban areas prior to 2010 but shifted to landing in both urban and rural areas thereafter. Consequently, the GOV recorded only a portion of the entire harvest for each year. Such discrepancies can lead to an unpredictable bias in the estimation and trend of stock size, as already noted by Omori et al. (2016).

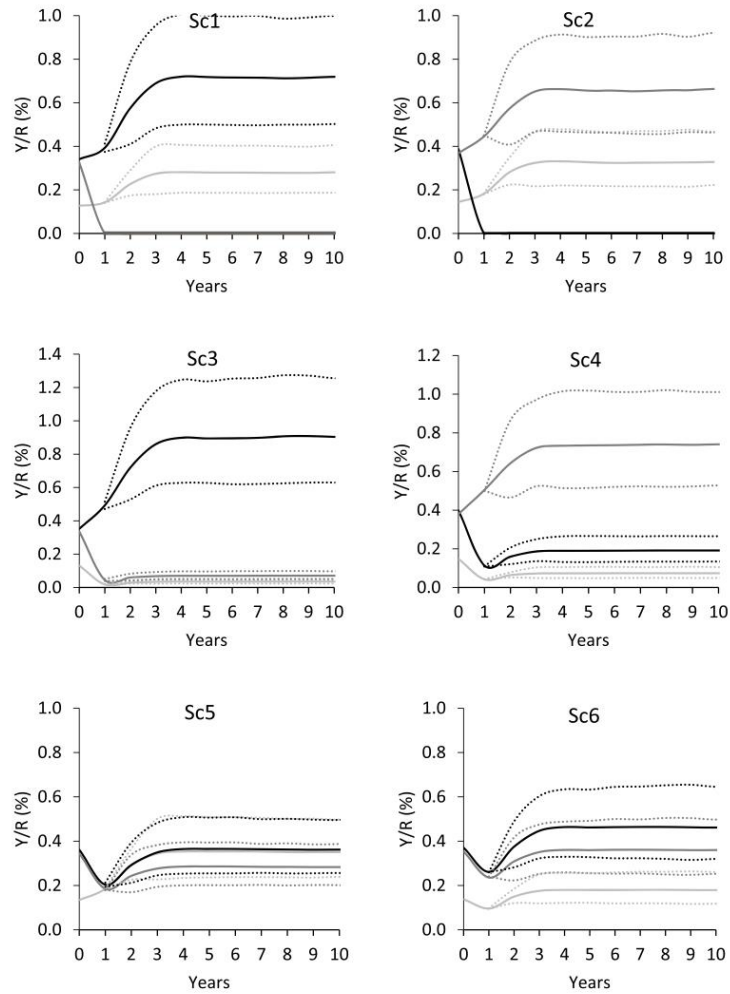
Our study shows evidence for a major change to the most used gear in the lane snapper fishery. Additionally, at least three different regimes of fishing effort have been identified since 1982 in the Colombian Caribbean (García, 2010; Manjarrés, 2004). Accordingly, we used several raising factors to separate the effects of port, fleet and period (i.e., month and year) on the reconstructed landings. To date, the reconstructions of marine landings in Colombia and other developing countries have used a single raising factor for all time periods (Lindop et al., 2015; Wielgus et al., 2010; Zeller and Pauly, 2007), thus ignoring differences in harvesting power among fleets and the effects of different fishing effort regimes on the reported landings. The reconstruction of world landings highlights the importance of small-scale fisheries for attaining resource sustainability and for providing food security. However, this reconstruction could lead to an incorrect estimate of fishing pressure and population status.

##### 4.2. Determining population status

Fishery model parameterization under a data-poor framework is tied to several underlying assumptions (Leonart and Salat, 1997; Prince et al., 2015; Rätz et al., 2010). The  $vBGP$  obtained from fishery-dependent data in this study are probably biased because of continued overfishing that leads to age truncation (Kenchington, 2013; Saari et al., 2014). Furthermore, the fishery-dependent length frequencies used to estimate the growth parameters underestimated the smallest and largest fish. Additionally, indirect estimators spread the bias from the  $vBGP$  in calculating natural mortality (Kenchington, 2013; Then et al., 2015), which is important because population status depends on  $M$ . This is of great concern because inclusion of random data led to contradictory information about population status.

Few studies have compared the performance of indirect estimators obtained from "known" natural mortality rates of fish (Gislason et al., 2010; Kenchington, 2013; Newman, 2002; Then et al., 2015). Most argue that estimators based on  $T_{max}$ , mainly Hoenig's estimator, should be selected instead of life-history correlation estimators (Kenchington, 2013; Then et al., 2015). Pauly's and Ralston's estimators, which are based on  $k$ , lead to higher  $M$  values (Kenchington, 2013). Additionally, Pauly's estimator proved to perform badly for species that exhibit fast initial growth and later move toward slow growth, such as snappers (Kenchington, 2013). Then et al. (2015) found that the predicted error in the estimation of  $M$  using Pauly-based models was twice that using Hoenig-based models.

We found that Pauly's estimator and similar estimators based on life-history correlations led to an underexploited or less exploited population status compared to Hoenig's estimators, meaning that current effort could still be increased. This seems unfeasible because fishing mortality of small-sized fish between early and later years tended to increase, while landings and  $CPUE$  have been strongly reduced. The annual  $CPUE$  of the encircling gillnet vessels, including all caught species, was ten and eleven times higher than those of the bottom gillnet and long-line vessels, respectively. During the study period, the annual-average catch per vessel fell by 20.40% for encircling gillnets, by 48.06% for bottom gillnets and by 75.27% for the long-line vessels. These results support our findings of a growing trend toward overexploitation of both species. When Gislason's  $M$ -at-size estimator was employed, variations in  $L_{\infty}$  and  $k$  led to large changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$  because both parameters play an important role in the estimation of natural mortality (Gislason et al., 2010). The lane snapper and white



**Fig. 6.** Simulation of lane snapper yield per recruit ( $Y/R$ ) by the long-line (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets. Scenarios (Sc) depending on management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment model and  $L_{\infty} = 64.94$  cm,  $K = 0.19 \text{ yr}^{-1}$  and  $M = 0.23 \text{ yr}^{-1}$ . All scenarios reached a relative  $SSB_{20\%}$  in a period of 8 years.

grunt, which are distributed around warm and temperate waters, exhibited quite different  $M$ -at-size values depending on the  $vBGP$ . The  $M$ -at-size estimator appears to perform poorly for species that exhibit large variations in  $L_{\infty}$  and  $k$ , so both uncertainty and bias associated with the calculation of  $vBGP$  have a strong impact on the estimation of  $M$ . Similarly, Abella's  $M$ -at-age estimator led to the greatest overexploitation status among the estimators tested. Abella's estimator appears to work well for temperate species with juveniles and adults spatially segregated (Abella et al., 1997), but underestimation of the  $M$  of tropical species with small and large individuals sharing the same areas could lead to overestimation of current fishery pressure. This would also lead to a large underestimation of fishing mortality in the long term (Clark, 1999).

The stock assessment is also conditioned by either the fast-growing or slow-growing hypothesis (Mellon et al., 2010). An increase in  $k$  means a reduction in longevity, affecting indirect estimates of natural mortality (ICES, 2014). Variation in the  $vBGP$  also led to large changes in the levels of  $F/F_{msy}$  and  $SSB/SSB_{max}$  for both species. Fast- and slow-growing have been assigned to the lane snapper and the white grunt distributed in the Caribbean Sea and East Atlantic Ocean (see Methods). However, great uncertainty remains in their estimates and comparisons because different methods and populations with diverse exploitation levels have supported the estimation of  $vBGP$ . Considering bias in the calculation of  $vBGP$  and  $M$  in our study, the stock assessments and management simulations were better resolved through testing both growth

hypotheses. Our analyses showed that a higher  $k$  for lane snapper produced greater changes in the overexploitation level throughout the study period. Although not totally reliable, this  $k$  easily tracked changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$ . Greater changes between years would be desirable, given that the responses of the studied populations to unexploited or slightly exploited pressures are unknown.

Rindorf et al. (2016) stated that species of smaller asymptotic length (e.g., herring) do not acquire an inherently precautionary  $F_{msy}$  range, as occurs in species with a larger  $L_{\infty}$  (e.g., cod). Additionally, these authors mentioned that the  $SSB$  of stocks with high asymptotic length are less prone to being affected by poor recruitment in some years. The reconstruction of virgin biomass and annual stock size obtained by our study was conditioned by 1) a steady state assumption (Rätz et al., 2010), 2) an assumption of no changes in the selectivity by gear between years, and 3) a possible strong year-class effect in some years (as occurred in 2010 with one exceptional reconstructed white grunt catch). These conditions, along with the uncertainty in the estimation of the  $vBGP$ , probably contributed to masking the total effect of a landings decrease on the trend of stock size.

Punt et al. (2011) outlined among-stock penalties for assessing data-poor stocks when they are jointly caught with rich-data stocks, given that rich-data stocks are insensitive to penalties. However, the precision of outputs for data-poor stocks is greater when among-stock penalties are taken into account. Most of the stocks of small-scale fisheries in developing countries are data poor, meaning that the abovementioned approach cannot be implemented. From a derivation of the concept of “borrowing” data, as suggested by other authors (Kokkalis et al., 2015; Prince et al., 2015), we contrasted the best available local information and stock-related data from the literature to choose the least biased life-history parameters.

Overall, the evidence suggests that higher than natural fishing mortality occurred for highly targeted fish sizes. This finding is in agreement with the suggestion of using a conservative natural mortality in a data-poor context (Clark, 1999). Therefore, the estimators of natural mortality based on life-history correlations, which produced high values, were dismissed (Kenchington, 2013; Then et al., 2015). Although it cannot be claimed that the best data were found, plausible biological parameters of both species in the Central Guajira were introduced into the assessment models. The selected natural mortality rates (from 0.13 to 0.24  $yr^{-1}$ ) produced overexploitation statuses for both species. Our findings indicate that the level of exploitation of stocks according to  $F/F_{msy}$  (e.g., 1 < overfishing < 3.5 for lane snapper; Fig. 5a) and  $SSB/SSB_{max}$  (e.g., overfished, 5–18% for white grunt; Fig. 5b) were sensitive to changes in the  $vBGP$ . However, the  $vBGP$  tested in this study did not lead to contradicting population statuses for either species, as did the different estimators of natural mortality.

The problems of what particular level of overfishing and what spawning stock biomass remain under the current fishing pressure cannot be entirely addressed. Uncertainty analyses are increasingly used for some data-rich fisheries to avoid selecting the “best” parameters and to rather incorporate all plausible combinations of them (Scott et al., 2016). To implement this approach in data-poor fisheries, the quality and quantity of basic length information must be improved. Reliable model input data are required to separate bias from uncertainty. We suggest that tracking population changes in the trends of  $F/F_{msy}$  and  $SSB/SSB_{max}$  is a safer way of advising data-poor fisheries. This would avoid the management of data-poor fisheries based on the uncertain estimation of stock biomass and under a comprehensive range of probable fishing mortalities.

The stock assessment for lane snapper performed with the GOV data produced a higher overexploitation status than when the GOV and PAR combined data were used. A large underestimation of landings in rural areas for long-line and gillnet fleets by the government

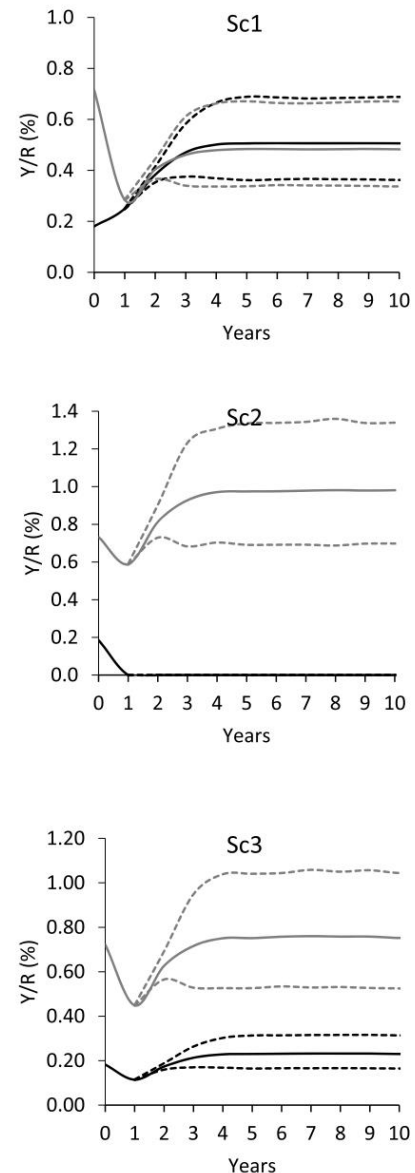


Fig. 7. Simulation of white grunt yield per recruit ( $Y/R$ ) by the longline (light grey), bottom gillnet (black) and encircling gillnet (dark grey) fleets. Scenarios (Sc) depend on management of fish size, deployed fleet and the food security and profits of fishing communities (see text). This analysis uses constant recruitment model and  $L_{\infty} = 41.65$  cm,  $K = 0.27$   $yr^{-1}$  and  $M = 0.15$   $yr^{-1}$ . All scenarios reached a relative  $SSB_{20\%}$  in a period of 8 years.

monitoring program yielded small variations in landings during the study period and provided a false view of the selectivity pattern for the lane snapper fishery. Managing this stock based on the government monitoring program increases the collapse risk of the lane snapper population, despite the fact that a higher over-exploitation status has been indicated. This occurs because a stock assessment indicating a large overexploitation matched with low changes in  $F/F_{msy}$  and  $SSB/SSB_{max}$  among years (Fig. 5) would mean that the population timeframe observed (2009–2014) was quite stable, despite the fact that the stock is greatly overfished and the fishing pressure is unsustainable. This can lead to the false assumption that the lane snapper population can better absorb the threats of overfishing. Conversely, by including data from participatory monitoring, we highlight that in nine years, the  $SSB/SSB_{max}$  could have been reduced by half while  $F/F_{msy}$  could have been doubled. Therefore, in data-poor fisheries, the overestimation of fishing mortality (Thorson and Prager, 2011) does not necessarily decrease the risk in management advice.

The stock assessment of the white grunt did not lead to distinct outputs of  $F/F_{msy}$  and  $SSB/SSB_{max}$ , regardless of which data source was used. This is because the encircling gillnet fleet dominated the species harvesting and the selectivity pattern for this fleet matched that of the gillnet fleet. Therefore, distinct landing points recorded by either monitoring program did not alter the selectivity pattern for the white grunt fishery. Nevertheless, the biomasses removed among fleets were quite different, implying that reaching a maximum sustainable yield ( $F/F_{msy} = 1$ ) would require that the biomass harvesting power by fleet be taken into account. This, however, is only feasible if landings recorded by the participatory monitoring program are considered in the stock assessment and further management simulations.

We found that the uncertainty associated with the different biological parameters and stock recruitment models used to perform each management scenario was moderate (lane snapper, 9%–33%; white grunt, 29%–40%). Consequently, the uncertainty identified during the definition of population status was considerably reduced when all the management scenarios were tested. According to our findings, the populations of lane snapper and white grunt are largely overexploited, meaning that recruitment may be compromised. The constant recruitment model included the underlying assumption of little or no effect of depleted stock size on recruitment (Brooks et al., 2010). However, it is also possible that the number of recruits per spawner increase at low population abundances (Jennings, 2001), with the possibility of a quasi-linear curve. Thus, irrespective of the actual stock-recruitment model and which life-history parameters correctly describe the two species, the management scenarios from a data-poor stock assessment could offer useful advice on a reasonable fishing effort reduction range (e.g., vessels per fleet). However, findings from stock assessment and management scenarios relate to the specific conditions of evaluation (Carruthers et al., 2014), so for providing management advice, the parameters of the model should remain constant over time.

#### 4.3. Conclusions

The government monitoring program led to a misleading description of landings and fishing mortality. This triggered riskier definitions of population status and of the overall fishery trend. The incorporation of data from the participatory monitoring program offered a better understanding of the exploitation patterns for the lane snapper and white grunt populations. Our findings suggest that sensitivity tests should become part of the advice for data-poor fishery management, as understanding the bias in the data for model parametrization cannot be overlooked. In addition, several limitations underlie the single-species stock assessment in a data-poor context. Some of them can be addressed (e.g., the definition of

population status), but others cannot (e.g., absolute values of  $SSB$ ). Therefore, control of fishing effort should be preferred for management instead of total allowable catch. Finally, our results support the idea that the decentralization of fishery management structures and the participation of local communities are required to obtain improved data. If stock assessments produce a more reasonable population status for key species, reliable advice can be offered for participatory management of small-scale fisheries in data-poor contexts.

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#### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.fishres.2017.01.015>.

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## 6.3. Annex 3. Parallel contributions

This section presents one scientific article and two book sections that were published during the thesis period and that contribute to the required framework to advise and manage the small-scale fisheries. However, these works are not included as part of the present thesis because do not accurately fill the main thesis goal.





## Perceived length at first maturity in the lane snapper, *Lutjanus synagris* (Linnaeus, 1758) (Perciformes: Lutjanidae), along the Caribbean coast of Colombia

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**Abstract.** Fishery management measures based on knowledge of fish reproductive biology involve estimation of indicators like length at first maturity, Lm. This indicator, as the case is for length-based indicators, is attractive due to its easy of calculation, in particular, in settings where sophisticated fishery assessment is not in place. On the basis of data originated in the artisanal fishery (2008-2014) estimation was made of length at first maturity of the lane snapper *Lutjanus synagris* in the Colombian Caribbean and the existence or not of spatial patterns in this indicator was investigated. Tests via resampling showed that length at first maturity vary significantly among regions in range of 16.9 cm to 28.9 cm, total length. Differences in gear (handlines and longlines, gillnets) regional size selectivity correlate with the Lm observed spatial pattern. It is hypothesised that fishing has gone so far as to alter the demography of *Lutjanus synagris* along the coast in terms of a feedback loop and concomitantly, has altered its length at first maturity, at least as perceived from the fishery data. A plea is made as to redefine management objectives, practice and research needs for the Colombian Caribbean artisanal fishery in view of new paradigm shifts proposals regarding life-history effects of selective fishing on the species concerned.

**Keywords:** Lane snapper, Colombian Caribbean, artisanal fishery, length at first maturity, selectivity

**Resumen.** Talla percibida de primera madurez en el pargo rayado, (*Lutjanus synagris*, Linnaeus, 1758) (Perciformes: Lutjanidae), en la costa Caribe de Colombia. Medidas de manejo pesquero basadas en conocimiento de la biología reproductiva de los peces incluyen la estimación de indicadores como la talla de primera madurez, Lm. Este indicador, como en general los indicadores basados en tallas, es atractivo por su fácil implementación, en particular, en escenarios donde aún no se cuenta con evaluaciones pesqueras sofisticadas. A partir de datos originados en la pesquería artesanal (2008-2014) se estimó la talla de primera madurez del pargo rayado *Lutjanus synagris* en la costa Caribe de Colombia y se buscó establecer la existencia o no de variación espacial en este indicador. Tests vía remuestreo mostraron que la talla de primera madurez varía significativamente entre regiones con un rango de 16.9 cm a 28.9 cm de longitud total. Dicha variación espacial correlaciona con características de selección de tallas de las artes usadas regionalmente (líneas de mano, palangres y redes agalleras). Se hipotetiza que en términos de un círculo de retroalimentación la pesquería ha alterado la demografía de *Lutjanus synagris* a lo largo de la costa y en consecuencia su talla de primera madurez, al menos según se percibe de los datos pesqueros. Se hace un llamado en el sentido de reformular los objetivos de manejo, la práctica y las necesidades de investigación en relación a

la pesquería artesanal del Caribe colombiano en vista de propuestas de cambio de paradigma en relación a efectos de la pesca selectiva sobre rasgos de la historia de vida de las especies afectadas.

**Palabras clave:** Pargo rayado, Caribe colombiano, pesca artesanal, talla de primera madurez, selectividad.

### Introduction

Several fishing management measures rest upon knowledge on the reproductive biology of fishes (e.g., minimum landing size). Hence, length at first maturity (from hereon, Lm) has been proposed as an indicator of the state of the fishery under the motto “let them spawn” with a target of 100% mature fish in the catch (Froese 2004). The conceptual easy of such approach is attractive in particular in settings where sophisticated fisheries assessment is not in place as the case still is in many tropical countries.

Artisanal fisheries at the Caribbean coast of Colombia are important in terms of food security and employment. Sadly, as the case is in many other places where resources are open access, the fishery is in a state of decline (García 2010) and in much need of administration if there is hope of restoration of its productivity (Saavedra-Díaz *et al.* 2015). In the search of easy to implement management measures the “Autoridad Nacional de Acuicultura y Pesca” (National Authority of Aquaculture and Fisheries), AUNAP (Spanish acronym), launched a campaign of recollection of maturity stages of landed fishes that covered most of the Caribbean coast of Colombia (MADR-IICA 2011, AUNAP-UNIMAGDALENA 2013, Fig. 1). This afforded the opportunity of addressing questions about potential spatial patterns of Lm of fishes in a tropical marine ecosystem and concomitantly, of assessing the current management practice of estimating an all valid Lm for the entire Caribbean coast of Colombia.

The lane snapper (*Lutjanus synagris*), taken as our case study, is (still) the most valued demersal fish targeted by the artisanal fishery in the Caribbean coast of Colombia (García 2010). This fish has supported heavy fishing for decades (Erhardt 1977). Various authors have proposed estimates of Lm for this fish, in particular, for the northern part of the Colombian Caribbean while for the southern part no antecedent was found (see Table I below). The present paper aims to establish and discuss spatial patterns in perceived (as data are fishery-dependent) Lm, i.e. the length at which 50% of the population become mature for the first time (Tsikliras *et al.* 2013), of *L. synagris* as a contribution to its

life-history and as input for its correct management.

### Materials and Methods

**Study area.** The Caribbean coast of Colombia has an extension of 1600 km approximately (Fig. 1). Situated well in the neotropic it harbours a variety of habitats: coral reefs, mangroves, sea grass beds, extended soft bottoms, etc., all of which are subject to fishing, even when in the interior of natural parks. The lane snapper is to be found in all these habitats and, although often signalled as coral reef representative, it is very well represented in soft bottoms (García and Ammenteras 2015). The Magdalena river, the main geographic accident, roughly divides the coast into two mayor areas: to the northeast of its delta (Magdalena-Atlántico and Guajira regions, Fig. 1) as an area subject to a more or less strong upwelling and low precipitation conditions (Andrade & Barton 2005), and the southwest as an area subject to notorious riverine input and high precipitation (all other regions, Fig. 1). Climate is bimodal with a rainy season and a dry season that take place in different months according to the location along the coast but with a tendency of dry months from December to April and rainy months the rest of the year.



**Figure 1.** Study area signalling the regions where lane snapper (*Lutjanus synagris*) maturity data were collected along the Caribbean coast of Colombia.

**Data bases.** Data correspond to the classification of measured individuals (total length, from hereon, TL;

all mentions of length in the paper refer to TL, under otherwise stated) caught by the artisanal fishery (mostly gillnets, hook and lines and longlines) into macroscopic defined maturation stages. The database assembled by AUNAP is divided into several areas: Guajira, Magdalena-Atlántico (merged here), Bolivar, Sucre and Cordoba that correspond to the political divisions (departments) in Colombia (Fig. 1).

This division was maintained, except for Magdalena-Atlántico, as they reflect different seascapes. Additionally two independent databases were included, one for Guajira originated in the work of the NGO ECOSFERA for comparative purposes, and another for the Parque Nacional Natural Corales del Rosario y San Bernardo (National Natural Park Corales del Rosario and San Bernardo, from hereon, San Bernardo NP; Jaimes 2011). Overall, the data covers the period 2008 to 2014 in monthly steeps.

#### Data processing

Data from the NGO ECOSFERA came divided into six progressive maturation stages (I, II, IIR, III, IV, V and VI), whereas data from AUNAP came divided into four maturation stages (Agudelo *et al.* 2011): I (virgin or immature), II (maturing), III (mature) and IV (spawned), thus the ECOSFERA data were constrained to fit the maturation stages in AUNAP data base by relabelling individuals in stages IIR, IV, V (just one individual) and VI as in stage IV. Data from San Bernardo NP came divided into five maturation stages (I, II, III, IV and V; Jaimes, 2001) but with just one individual in stage V that was counted then as in stage IV.

Individuals labelled as "indeterminate" in AUNAP database were excluded from analysis as they represented fishes whose maturation stage was not evaluated. For individuals with total weight but not total length this was estimated by means of the following equation:  $TL = (W/0.037)^{1/2.677}$ , where TL stands for total length and W stands for weight, obtained by fitting existing length-weight data to a linear model to estimate the parameters and then solving for TL (n=4780 data pairs).

Length data and their corresponding maturation stage were organized in 2 cm length classes. Sexually mature individuals, i.e., individuals that potentially have reproduced at least one time, were taken as those in mature stages III and IV. Regional values of Lm were estimated by means of a linearized logistic curve fitted to the proportion of sexually mature individuals by length class (the mean of the length class; length classes with less

than 10 individuals were not considered) adjusted by the maximum proportion of mature fishes in the sample so that that length class represented 100% mature individuals (King 2007). In order to generate confidence intervals and statistically compare the regional Lm an ad hoc procedure was use as follows: The x-y pairs of the original regression were resampled with replacement 1000 times, each time outputting an intercept and slope. For each round, an Lm was calculated as the ratio between intercept and slope with sign changed of the linear regression (King 2007). The mean value of the 1000 resamples is taken as the most representative Lm for the samples. Bias corrected and accelerated 95% confidence intervals were fitted to the mean Lm of the 1000 resamples.

A statistical comparison was conducted between the Lm estimated as above for each pair of regions. The resamples were shuffled together 1000 times and the difference in resampled means recorded and compared with the observed difference in means. The percentage of times the resampled differences were larger or equal than the observed difference was taken as the p value with significant difference at  $P < 0.05$ . The above calculations were conducted with a demo version of the Excel add-in application Resampling Stats, to be found at <http://www.resample.com/download-excel/>.

The hypothesis that the length distribution and mean length between all possible pairs of regions was the same was tested by means of a Kolmogorov-Smirnov test and a t-test, respectively, both based on permutations (10000 permutations) as implemented in the program PAST (Hammer *et al.* 2001).

#### Results

Table I shows previous Lm estimates for *L. synagris* in the Colombian Caribbean. The southern Colombian Caribbean has not receive much attention in this respect (compare Table I with Fig. 1) and estimates are heterogeneous. Table II shows a basic description of the database. Data are rather heterogeneous in numbers and mean sizes. Mean length of landed fishes between regions was significantly different ( $P < 0.05$ ) in all pairwise comparisons excepting the comparison San Bernardo NP vs. Cordoba AUNAP ( $P > 0.05$ ). Length distribution is statistically different between regions (Kolmogorov-Smimov test,  $P < 0.05$ ) with the exception again of San Bernardo NP vs. Cordoba AUNAP (Kolmogorov-Smirnov test,  $P = 0.05$ ).

**Table I.** Previous estimates of length at first maturity (Lm) of *Lutjanus synagris* for the Colombian Caribbean. TL is total length.

Source	Spatial coverage	Time coverage	Data type	Stages	Lm (TL, cm, both sexes)
Posada-Peláez <i>et al.</i> (2012)	Magdalena	2008-2010	Artisanal fishery	5, mature: IIR, III to V	28,0 <sup>1</sup>
Posada-Peláez <i>et al.</i> (2012)	Magdalena	2008-2010	Artisanal fishery	5, mature: IIR, III to V	31,6
Arteaga <i>et al.</i> (2004)	La Guajira	2000-2001	Artisanal Fishery	6, mature: IIR, III to V	30,1
Arteaga <i>et al.</i> (2004)	Tayrona Park (Magdalena)	2000-2001	Artisanal fishery	6, mature: IIR, III to V	34,0
Rodriguez <i>et al.</i> (1999)	Colombian Caribbean	1995-1996	Demersal survey	5, mature: III to V	26,9
Barreto and Borda (2008)	Colombian Caribbean	2006-2007	Artisanal fishery	4, mature: II to IV	29,3
Rodriguez and Páramo (2012)	La Guajira	2006	Demersal survey	5, mature: III to V	21,4 <sup>1,**</sup>
Altamar <i>et al.</i> (2015)	La Guajira	2007-2014	Artisanal fishery	Not stated	25,1 <sup>2</sup>

<sup>1</sup>Estimated via maximum likelihood. <sup>2</sup>Estimated via generalized linear models, otherwise estimations based on linearized version of logistic model. \*\*For females only.

**Table II.** Basic description of the data bases used in investigating spatial patterns of length at first maturation (Lm) of the lane snapper (*Lutjanus synagris*) in the Caribbean coast of Colombia. Length of fish is total length (TL). Confidence intervals (CI) are bias corrected and accelerated 95 % confidence intervals.

Region	N	MinTL (cm)	MaxTL (cm)	MeanTL (cm)	CIinf (cm)	CI sup (cm)
Guajira ECOSFERA <sup>1</sup>	697	14.4	56.0	28.8	28.4	29.2
Guajira AUNAP	1035	19.3	44.6	28.0	27.7	28.2
Magdalena-Atlántico AUNAP	619	18.6	54.0	30.5	30.1	30.9
Bolivar AUNAP	259	17.0	41.0	26.2	25.6	26.8
San Bernardo NP <sup>2</sup>	826	12.2	36.2	21.5	21.2	21.8
Sucre AUNAP	1588	11.6	44.5	19.4	19.2	19.6
Cordoba AUNAP	98	13.0	35.2	22.0	21.0	23.0

<sup>1</sup>Temporal coverage goes from 2010 to 2011. <sup>2</sup>Temporal coverage goes from 2008 to 2009. For all other regions, (AUNAP) temporal coverage goes from 2012 to 2014.

Table III shows the estimated Lm for each region. Values are heterogeneous. AUNAP data suggest a parabolic change of Lm as one moves to the south with maxima in Bolivar (Table III; Fig. 1). ECOSFERA and AUNAP data yield fairly different Lm estimates for Guajira (Table III) that turned out to be significantly different (P<0.05). Lm for Guajira ECOSFERA, Magdalena-Atlántico AUNAP and Bolivar AUNAP were not different statistically among each other (P > 0.05) as well as Bolivar AUNAP and San Bernardo NP (P > 0.05), whereas all other Lm comparisons were significant (P < 0.05) which suggest a decreasing gradient in Lm to the south.

The mechanistic explanation of the different Lm among regions is shown in Fig. 2. For a given maturation stage its mean length is quite different among regions. Thus, for instance, a fish of size 20

cm in Cordoba AUNAP, where it is included as in stage IV, would probably be regarded as in stage I in the other regions, except Sucre AUNAP (Fig. 2).

**Discussion**

The lane snapper appears to reach maturity (Lm) at different mean sizes along the Caribbean coast of Colombia. Spatial variation in Lm are not rare in particular for well distributed species like *L. synagris*, e.g., Stahl & Kruse (2008), Head *et al.* (2014). However, the data in this study are fishery dependent and thus, how accurately they represent what is actually happening in the water is an open question.

Possible explanations of the patterns seen here include (1) different gears and gear selectivity among regions such that the proportion on mature fish per size class is perceived as different,

**Table III.** Estimation of length at first maturity (Lm) of the lane snapper (*Lutjanus synagris*) per region in the Caribbean coast of Colombia. Lm is the mean Lm value of 1000 resampling rounds of the linearized logistic curve fitted to the data (see Materials and Methods) and refers to total length. Confidence intervals (CI) are bias corrected and accelerated 95 % confidence intervals.

Region	Lm (cm)	CIinf (cm)	CI sup (cm)
Guajira ECOSFERA	27.4	27.2	27.6
Guajira AUNAP	22.3	22.2	22.4
Magdalena-Atlántico AUNAP	27.6	27.0	28.0
Bolivar AUNAP	28.9	23.8	31.4
San Bernardo NP	25.9	25.8	26.0
Sucre AUNAP	16.9	16.8	17.0
Cordoba AUNAP	17.3	17.2	17.4

(2) actual existence of different stocks each with its own peculiar demography, (3) marked inconsistency in the observers (different teams for each region) when classifying individual fish into stages of the reproductive cycle, (4) bias due to different sampling sizes.

Incidence of fishing gears used in the regions is shown in Table IV. In all regions the bulk of gears are hooks (hook and line and longlines) and, secondarily, gillnets, except in Guajira, thus, differences in gear type can be counted out. On the other hand, each region is idiosyncratic in terms of the size, i.e., selectivity, of gears used (see Table V) and hence, the differences in landed fish size structure (Kolmorov-Smimov test), mean TL and mean Lm among regions (pairwise tests) most likely respond to such features.

Support for this assertion comes from the finding that regional mean Lm positively responds to regional mean TL (Fig. 3), and regional mean TL, in turn, responds to regional hook size (negatively as scale is inverse, mean size category as defined by the manufacturer) and gillnet mesh size (positively, Figs. 4 and 5, respectively). In summary, regions such as Sucre AUNAP and Cordoba AUNAP with the lowest Lm (Table III) and low to no incidence of gillnets (Table V) concomitantly show the smallest hooks (Table V), which suggests that the fishery has been adapting to progressively smaller fish as decimation of bigger fish took place. The effect of fishing in the demography of exploited fish species is well known. In terms of reproduction, fish mature early, i.e., at a smaller size and younger age because gear selectivity and heavy fishing mortality favours early manifesting traits (Sharpe & Hendry 2009, Enger *et al.* 2012, Audzijonyte *et al.* 2013).

Hooks and gillnets produce bell-shaped selectivity curves that provide refuge for fishes above and below the catchable size range. This

feature has been postulated as a partial protection against the so-called fisheries-induced evolution (Hutchings 2009, Jorgensen *et al.* 2009). Nevertheless, Jorgensen *et al.* (2009) do recognize that once fishing mortality reach a certain point maturity age drops.

By far predominant gears in the artisanal fishery in Colombian Caribbean are hooks and gillnets. The observed spatial pattern of Lm would then suggest that southern stocks (San Bernardo, Sucre, Cordoba) are in very bad condition compared to the northern stocks and that fishery selection has probably gone so far as to change their genetic make-up.

The hypothesis that the observed spatial pattern in Lm originates in the existence of different stocks with differential demography, cannot be totally dismissed, however. To our knowledge two studies exist on stock differentiation of *L. synagris* in the Colombian Caribbean. One, based on morphometric and meristic data (Gómez-Canchong *et al.* 2004) and the other based on microsatellite-type molecular markers (Landinez-García *et al.* 2009). These studies identified three distinct stocks: Guajira-Rosario Archipelago (that includes San Bernardo NP), Santa Marta (intermediary between Guajira and Magdalena-Atlántico regions) and Capurgana (outside our study area, Fig. 1). Thus, there is some indication that detected differences in size structure in the fishery may have a correspondence in population genetics of *L. synagris* along the Caribbean coast of Colombia. The finding that mean TL and size structure of San Bernardo NP and Cordoba AUNAP are not statistically different, but that Lm is, further supports the reality of different stocks of *L. synagris* in the Caribbean coast of Colombia. Nevertheless, further detailed and extended studies are needed to settle this point.

**Table IV.** Incidence (percentage) of gear type used by the artisanal fishery in the Caribbean coast of Colombia in capturing lane snapper (*Lutjanus synagris*). Hooks include hook and line and longlines. Gillnets include fixed and drifting gillnets and beach seines. A marginal number of fish traps and cast nets are included in the database.

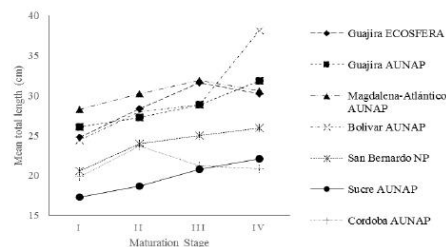
Region	Hooks	Gillnets	Skin diving	Traps	Cast nets
Guajira ECOSFERA	40.1	59.6	< 1	0	0
Guajira AUNAP	3.4	95.3	0	1.3	0
Magdalena-Atlántico AUNAP	81.8	16.6	0	1.6	0
Bolívar AUNAP	79.2	20.8	0	0	0
San Bernardo NP	98.3	1.7	0	0	0
Sucre AUNAP	99.8	0	0	0	< 1
Córdoba AUNAP	80.0	19.0	0	0	1

**Table V.** Size characterization of hooks and gillnets used by the artisanal fishery in the Caribbean coast of Colombia in capturing lane snapper (*Lutjanus synagris*). Hooks size according to size categories as given by the manufacturer, in inverse scale with respect to its actual size; gillnet size is mesh size in inches. NG= not given. NDB= not in database.

Region	Hooks	Gillnets (mesh size)
Guajira ECOSFERA	8.6	3.3
Guajira AUNAP	7.6	3.5
Magdalena-Atlántico AUNAP	9.5	2.3
Bolívar AUNAP	10.9	2.8
San Bernardo NP	NG	NG
Sucre AUNAP	11.4	NDB
Córdoba AUNAP	11.9	2.7

One feature that warrants explanation is the difference in Lm between Guajira ECOSFERA and Guajira AUNAP (Table III). Both region and time period of origin of the data are similar and so are their mean lengths (Table II), although statistically different (ICs in Table II). On the other hand, size structure is different (Kolmogorov-Smirnov test,  $P < 0.05$ ) which would be the mechanistic explanation for the difference in Lm.

Hammer 2001) confirms that fishes of stages III and IV (as constrained here) in ECOSFERA data are significantly bigger than fishes in the same stages in AUNAP data ( $P < 0.05$ ). The ECOSFERA data includes more landing places in the region and broader fishing deep (J. Ramirez unpublished data) than AUNAP data that rather concentrates in urban settlements. This would bring bigger fish to port, as it is common pattern that bigger (probably) mature fish predominates in deeper waters.

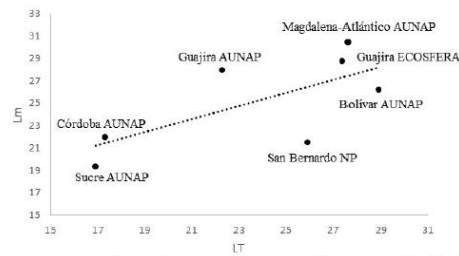


**Figure 2.** Mean size (total length) per maturity stage and region of the lane snapper (*Lutjanus synagris*) as landed by the artisanal fishery at the Caribbean coast of Colombia.

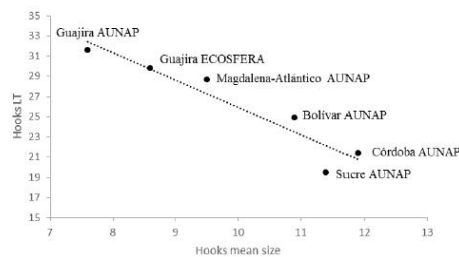
In summary, the emerging scenario appears to be one in which heavy fishing conducive to progressively smaller landed fish as attested by fishermen (García 2010), occurs in (and interacts with) a background of differentiable stocks of *L. synagris* as the most plausible explanation for the correlation between varying regional Lm and selection characteristics of regional fishing gears.

The incidence of gillnets with a smaller mesh size in Guajira AUNAP, which would be conducive to smaller fish landed and hence smaller Lm, is much higher compared to Guajira ECOSFERA (Tables IV and V). A t-test based in 10000 permutations (PAST,

In which extend sample size and evaluation teams differences influence the results is an open question. The number of fishes looked at in the regions goes to the hundredths and even to the thousands, thus sample size does not appear to be a problem. Estimation of bias caused by different teams of evaluators would require a process of field validation that has not taken place. However, in view of the constrains imposed by the size range of the landed fishes and the selectivity characteristics of the fishing gears, it appears unlikely that the observed spatial pattern in Lm be just the result of



**Figure 3.** Relation between length at first maturity (Lm) and mean total length (TL) in the lane snapper (*Lutjanus synagris*) fishery across regions in the Caribbean coast of Colombia.

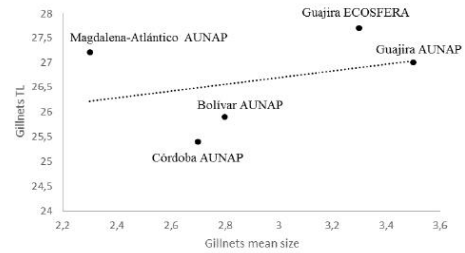


**Figure 4.** Relation between mean total length (TL) for hooks and hook scale (inverse scale, mean size category as defined by the manufacturer) in the lane snapper (*Lutjanus synagris*) fishery across regions in the Caribbean coast of Colombia. Note that San Bernardo NP database does not report on hook size.

differences between evaluators.

The estimation of Lm is quite sensible to the procedures in calculating it, even the decision of employing the lower, upper or mean size class point when fitting size at maturity data to a linearized logistic regression alters the result. Table I testifies of the variety of previous Lm estimates for *L. synagris* in the Colombian Caribbean, that are, however difficult to compare with the results here or among them as technical but crucial details are lacking, not to mention that the number of maturity stages, the stages considered mature and the estimation method differ among studies (Table I).

Lessons to be learned from this exercise concerning the management of this important species for the artisanal fishery in Colombian Caribbean include: (1) it is wrong to postulate an all valid single Lm for *L. synagris* covering the Colombian Caribbean by pooling maturity data from such different places, as the practice has been so far, e.g., Barreto and Borda (2008),



**Figure 5.** Relation between mean total length (TL in cm) for gillnets and gillnet mean mesh size in lane snapper (*Lutjanus synagris*) fishery across regions in the Caribbean coast of Colombia. Note that San Bernardo NP database does not report on gillnet size. In Sucre AUNAP no fishing is done with gillnets at least according to the AUNAP database.

AUNAP-UNIMAGDALENA (2013); (2) the target of 100% mature fish in the catch should probably be abandoned as it would exacerbate the truncation of natural size distribution structure and hence increase the risk of fishing induced evolution that might be irreversible (Stenseth & Rouyer 2008, Marty *et al.* 2015); (3) in future campaigns of gathering biological data more attention should be paid to the design of field work so as to achieve standardization of sampling effort and coverage, in particular, the depth range affected by the fishery should be properly covered; (4) intercalibration exercises among evaluation groups should be conducted before field work and (5) an analysis protocol for fish maturity data should be agreed upon by researchers as to allow comparison of studies.

In view of the new proposed paradigm shift regarding selectivity in fisheries that postulates that in the context of ecosystem-based fisheries management (EBFM) a more appropriate approach would be to strive towards balanced harvesting meaning distributing fishing mortality across trophic levels such that ecosystem properties like size-spectrum slopes and biodiversity be conserved, i.e., away from excessive selectivity (Zhou *et al.* 2010, Rochet *et al.* 2011, Garcia *et al.* 2012), a profound reflexion and reconsideration of objectives (and associated research needs) regarding the management of Colombian Caribbean artisanal fisheries is necessary, if our seas are to continue providing food security and employment.

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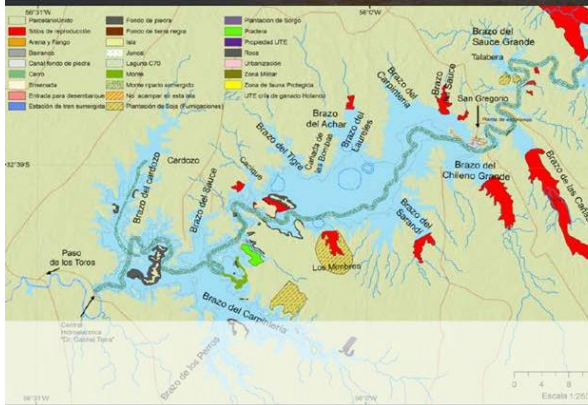
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# Fishers' knowledge and the ecosystem approach to fisheries

Applications, experiences and lessons in Latin America



## Diversity of small-scale fisheries and fishery agreements from the participatory management perspective in Colombia

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### ABSTRACT

Small-scale fisheries in Colombia occur in diverse areas including river basins, estuaries, and coastlines, for both consumption-based and ornamental fisheries, and carried out by mestizo, indigenous and afro-descendant communities. These fisheries are overexploited or approach maximum sustainable yield. Meanwhile, government capacity and policies for fisheries management are weak and many fishing communities have reached a state of poverty. These conditions have generated an increased interest in the implementation of participatory fisheries management by artisanal fishermen, government and other stakeholders. We analyze 36 fisheries agreements established during the last two decades using similarity analysis, nonparametric multidimensional scaling and similarity percentages to distinguish fisheries agreements according to different management measures proposed by fishermen. Fishermen recognized similar problems despite differences in ethnicity. However, fishermen gave different priority to fisheries measures and activities to carry out their specific fisheries agreements, depending on target species. Differences in fisheries agreements were found between consumption-based and ornamental fishing ( $P < 0.05$ ), between river basin and marine fisheries ( $P < 0.05$ ), and among fisheries in lakes, lagoons, and reservoirs ( $P < 0.05$ ). Small-scale fisheries in Colombia share several fisheries conditions, but these all show differences according to the social perspective of each community to solve its specific fisheries problems. Currently, small-scale fisheries in Colombia face important challenges associated with their diversity/heterogeneity, the poor government capacity for fisheries management, the enforcement of participatory management by different stakeholders (mainly NGOs) with different approaches that respond to their own goals.

### INTRODUCTION

Most of the world's fisheries are considered small-scale; however, efforts towards funding management and research have largely targeted industrial fisheries (McConney and Charles, 2008). This has led to the management of small-scale fisheries under conventional approaches, focusing on single-species fisheries with little to no consideration of the human dimension (Pomeroy, 1995) and disregarding the socio-economic needs of communities and the potential benefits of participatory governance (Berkes *et al.*, 2001).

Governments often consider community participation when there is dissatisfaction regarding the use of natural resources or to decentralize management, while encouraging the empowerment among communities (McGrath *et al.*, 2008, Pomeroy and Rivera - Guieb, 2006). This has led to a greater interaction between the State, resource users and other stakeholders in a concerted effort to solve problems and create social opportunities for the fisheries sector (Hartoto *et al.*, 2009; McConney and Charles, 2008; Symes, 2006).

Participatory management proposes shared responsibilities (Berkes *et al.*, 1991), rights and duties among the government, local users of natural resources and all parties involved in a fishery (Berkes, 2008; Carlsson and Berkes 2005, Berkes *et al.*, 2001; Pomeroy, 1995). Nonetheless, the implementation of shared management or Co-Management is complex, as the same resource can be subject to different forms of management by distinct government agencies and hold different interests within communities.

At the same time, fishing communities are heterogeneous entities that undergo constant change, implying that co-management is a state of continuous problem-solving rather than a static condition (Carlsson and Berkes, 2005). As such, Co-Management should be adaptive (Armitage *et al.*, 2007) as fishing communities tend towards resilience, given their ability to learn from previous experiences (Berkes, 2008; Salas *et al.*, 2004) and to cope with unexpected results (McConney and Charles, 2008). Nonetheless, the dynamics associated with small-scale fisheries management call for higher levels of administration, since in many cases these levels are required beyond the community level to solve the problems that need to be addressed (Nielsen *et al.*, 2004). In this sense, the involvement of small-scale fishing communities in decision-making in Colombia is precarious, and only within the past few years has there been a clear interest on behalf of the government and the private sector towards the implementation of a participatory management approach. However, the Colombian government is facing several challenges in developing this type of management in the country.

Colombia, located in the equatorial zone of South America, has many marine and inland water environments where indigenous, Afro-descendant and mestizo communities develop small-scale fisheries using different methods on a variety of resources (Rueda *et al.*, 2010; Lasso and Morales, 2011). Additionally, almost 90% of fisheries have reached or exceeded the maximum sustainable yield, under a governmental management scheme that is incipient and unstable (Wielgus *et al.*, 2009; Gutierrez, 2010).

In order to generate baseline information for formulating legislation that provides a basis for participatory management as a formal management mechanism for small-scale fisheries in Colombia, we collected fisheries agreements elaborated in different parts of the country, all of which apply to different water bodies (marine or inland) and which comprise the participation of different ethnic groups. This paper examines and analyzes the categories of existent fisheries agreements as well as the actions for implementation proposed by fishermen within participatory processes, to identify local problems and potential solutions.

#### **METHODS**

In Colombia were identified many actions and participatory process for fishery resources management. However, just 36 of these presented into their respective public documents details about actions and participatory measures for management of fishery resources. Information was obtained from 36 participatory processes (participatory diagnosis, formulation of agreements, fishing agreements or administrative decision by the national fisheries authority) to define local fishing agreements carried out by the own communities, these helped by different stakeholders and during the last 20 years.

Each participatory process which led to some form of fishing agreement, could or not take into account the participation of the fishing national authority. In many cases the fishing agreement just involved the community and collaborating organizations like

NGOs, while in other cases, the national fishing authority was directly involved in the fishing agreements generation. In Colombia only can exercise like fishing authority the government agency created with this specific purpose, however under the current national law the local communities can be organized for finding local solutions to their own problems. This way is including the possibility of producing fishing agreements involving local management. This implies that the process carried out for obtaining each fishing agreement could be different regarding other one. For this reason the agreements considered here were restricted to those where the information leading to the definition of fisheries agreements was produced with the participation of local communities.

Each particular participatory process used in this study produced several measures for implementing and complying the specific fishing agreements, which were published like legal or informative documents (Table 1). In this sense, all measures proposed in the different fishing agreements were extracted from each document and subsequently, these were clustered in agreement categories regarding the particular focus of each one (Table 2).

We created a matrix of presence/absence data that included the agreement categories and the specific implementation actions (fishing measures) proposed by each fishing community. It incorporated as many categories of agreement and actions for implementation as were recorded in the different participatory processes. No categories of agreement or fishery measures were considered in the analysis, unless they had been explicitly described (Table 2).

TABLE 1  
Fishery agreements proposed by fishermen in Colombia

State	Place	Spatial approach	Ethnic group	Water body	Communities included	Autor
Amazonas	Tarapoto lakes	Community	Indigenous	Lake-Lagoon-reservoirs group LLR	2	(Trujillo y Trujillo, 2009)
Amazonas	Yahuaraca lagoon system	State	Indigenous	Lake-Lagoon-reservoirs group LLR	1	(Agudelo <i>et al.</i> , 2013)
Amazonas	Mid and low basin of Caquetá river	State	Indigenous and Mestizo	Lake-Lagoon-reservoirs group LLR	2	(AUNAP, 2012b)
Atlántico-Bolívar	Totumo swamp	State	Mestizo	Swamp	1	(Niño, 2011)
Bolívar	Northwest side of Mompox Island	Town	Mestizo	Swamp	1	(Barrero <i>et al.</i> , 2010)
Bolívar	Magangué swamp	Town	Mestizo	Swamp	1	(Niño <i>et al.</i> , 2013)
Caldas	Amani reservoirs	Town	Mestizo	Lake-Lagoon-reservoirs group LLR	1	(Fundación Humedales - AUNAP, 2013)
Caldas	Low basin of La Miel river	Town	Mestizo	River	1	(Trujillo <i>et al.</i> , 2011)
Choco	Bahía Solano Littoral	Town	Afro-descendants	Coastlines	1	(Red de Frio, 2012)
Choco	Artisanal fishing exclusive zone	Town	Afro-descendants	Coastlines	1	(AUNAP, 2013)
Choco, Valle del Cauca, Cauca y Nariño	Pacific littoral	State	Afro-descendants	Coastlines	1	(Delgado <i>et al.</i> , 2010)
Córdoba	Urrá reservoirs	Town	Mestizo	Lake-Lagoon-reservoirs group LLR	1	(Fundación Humedales - AUNAP, 2012)
Cundinamarca	Tominé reservoirs	Town	Mestizo	Lake-Lagoon-reservoirs group LLR	1	(AUNAP, 2012a)
Guanía	fluvial start of Inirida	Community	Indigenous	River	7	(Zuluaga & Franco-Jaramillo, 2013)
La Guajira	Mid and High Guajira	State	Indigenous	Coastlines	9	Fundación Ecosfera - AUNAP, 2013)
Nariño	Tumaco cove	Town	Afro-descendants	Coastlines	4	(López <i>et al.</i> , 2006)
Nariño	Sanquianga National Natural Park	National Park	Afro-descendants	Coastlines	1	(PNN, 2009)

TABLE 2  
**Categories and fishery measures from fishery agreements proposed by fishermen in Colombia**

Agreement categories	Fishery measures	Agreement categories	Fishery measures
Biological controls	According to sex According to size spawning seasons Target-species fishing	Conservation	Prohibited areas for fishing Habitat alteration Repopulation Species saving
Fishing gear controls	Prohibited fishing gear Fishing gear specifications Fishing Aggregating Device (FADs)	Non-fishing activity controls	Education Looging Gold mining
Fishing type controls	Consumption Sport fishing Trade Ornamental fishing	Fleet controls	Aquaculture Agriculture Tourism Type of boat or engine
Temporary access controls	Closure period Specific fishing days		Permitted routes Travel speed
Area access controls	Regarding to others communities Specific fishing areas Rotation of fishing areas	Fishing strengthening	Fishing promotion Entrepreneurship Fishing port financial support
Trade controls	Prices Marketing Fishing buyer Fair trade	Government controls	Training Alternative fishing targets Good fishing practices Good fishing practices
Catch controls	Catch limits Number of fishery economic units Number of fishery gear units	Employment alternatives	Law compliance Non-fishing programs Women participation
Research	Conservation research Fishery research	Community strengthening	Strengthening of local organizations Implementation of local roles

From the presence/absence matrix, a quantitative matrix was constructed according to the number of actions proposed by fishermen for the implementation of each category of agreement identified. This allowed for the quantitative comparison under the same parameters for each of the 36 agreement proposals or fisheries agreements already in place. It was considered that information taken from the all agreements, entirely or in greater part, represented the interests and particular perspectives of the communities, as otherwise they would not have been proposed or approved by the fishing communities.

Data were analyzed using descriptive statistics to describe the trend associated with the proposals made by the fishermen regarding categories and fishery measures in fishery agreements. A management analysis using nonparametric multidimensional scaling (NMDS) was used to correlate the similarities between the categories of agreement proposed, taking into account the number of actions selected by each community to implement each category of agreement (Clarke and Warwick, 2001). Each community was assigned a category for factors; water body where the activity occurs (rivers, coastal marine, swamp and lakes, ponds, dam) basin associated to the fishery (Caribbean, Pacific and inland waters), category of catch (fish for consumption or ornamental fish) and type of aquatic environment where they develop fisheries (marine or inland waters).

To determine the factor level variation, the Bray-Curtis similarity index was used (Clarke and Warwick, 2001). The data was processed using fourth root to reduce the importance of fisheries agreements that were proposed with very low total references for the set of fishing communities studied; subsequently the results were plotted in two dimensions to the final configuration representing the lowest stress for 999 iterations. A one-way analysis of similarity (ANOSIM) was used to determine differences for the categories of each factor.

In addition to the significance level of the test, ANOSIM adds an R value to the degree of separation in terms of categories of fishing agreements ranging from 0-1. R values > 0.75 show good separation for the factors evaluated, R values < 0.50 show overlapping but with differentiation, and R values < 0.25 show little or undetectable separation (Wilhelmsson., 2006). Finally, similarity percentages (SIMPER) were used to identify categories of agreement that contributed to the differentiation of factors with R > 0.25. Only agreement categories with percentage of contributions greater than 5% were considered in the SIMPER analysis to avoid interference by outliers (Burt *et al.*, 2011).

## RESULTS

Of the 36 participatory process analyzed, fishing communities proposed 48 fishing measures that were grouped into 15 agreement categories (Table 2). We found that the distribution of agreements was not homogenous, but rather agreements largely grouped into 4-8 categories (Figure 1). Fishing communities of distinct regions differed in their need for agreements related to the exercise of control, conservation, development or fishing strengthening. Thereby, the categories of fishing gear controls, biological controls, conservation actions and fishing strengthening represented 50% of all agreement categories proposed by communities (Figure 2).

Taking into account the number of fishing measures within participatory process proposed by each community, different number of these were also found among communities (Figure 1). Some communities proposed approximately six measures in their fishery agreements (26%), while others considered between 9 and 11 fisheries measures (44%). Nonetheless, some participatory process proposed only two fishing measures while others reached 20 measures (Figure 1).

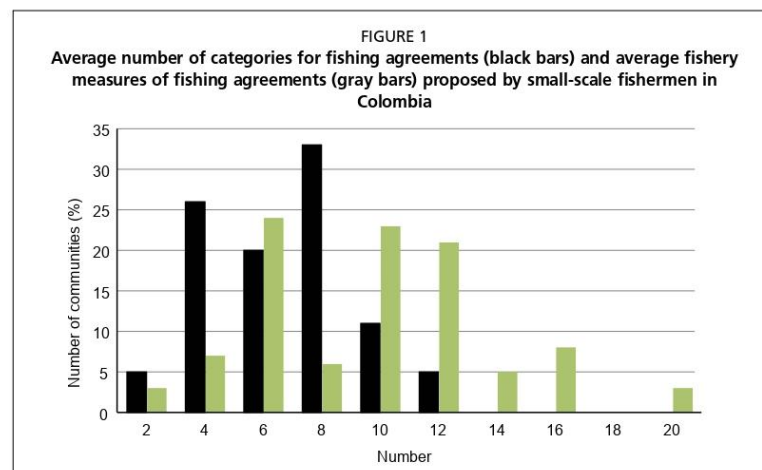


TABLE 3  
One-way similarity analysis for the factors basins, ethnic groups, water bodies, type of aquatic environmental and type of fishing resource

Factor	Comparison	R Value	Significance
Tipo de ambiente acuático	Global (Aguas continentales y aguas marinas)	0.313	0.001
Tipo de recurso pesquero	Global (Ornamentales y de consumo)	0.57	0.001
Cuenca	Global	0.301	0.001
	Aguas continentales y océano Pacífico	0.274	0.005
	Aguas continentales y mar Caribe	0.361	0.002
	océano Pacífico y mar Caribe	0.428	0.001
Cuerpos de agua	Global	0.350	0.001
	Ríos y Lagos-lagunas-embalses	0.297	0.003
	Ríos y zonas costeras	0.609	0.001
	Ríos y cienagas	0.164	0.152
	Lagos-lagunas-embalses y zonas costeras	0.325	0.004
	Lagos-lagunas-embalses y embalses	0.627	0.008
	Zonas costeras y cienagas	-0.145	0.719
Grupos étnicos	Global	0.054	0.217
	Indígenas y mestizo	-0.019	0.537
	Indígenas y afrodescendientes	0.081	0.161
	Mestizo y afrodescendientes	0.231	0.005

Differences between fisheries agreement categories proposed for consumption-based fisheries versus ornamental fisheries (Table 3) were found ( $P= 0.001$ , ANOSIM). Consumption-base fisheries incorporated agreements pertaining to categories such as biological controls, fishing gear controls, area access controls and conservation issues while communities working with ornamental fisheries did not (Table 4). On the other hand, for the case of ornamental fishing agreements proposals did not consider fishing strengthening, fleet controls, or government controls, as was the case of consumption-based fisheries. Differences between the categories of agreements proposed for freshwater fisheries and marine fisheries were also found ( $P= 0.001$ , ANOSIM). These differences were related to fisheries agreement categories such as fisheries promotion and research (scientific and traditional knowledge) for marine fisheries, whereas categories of agreements such as temporal access controls and area access controls were more important for freshwater fisheries.

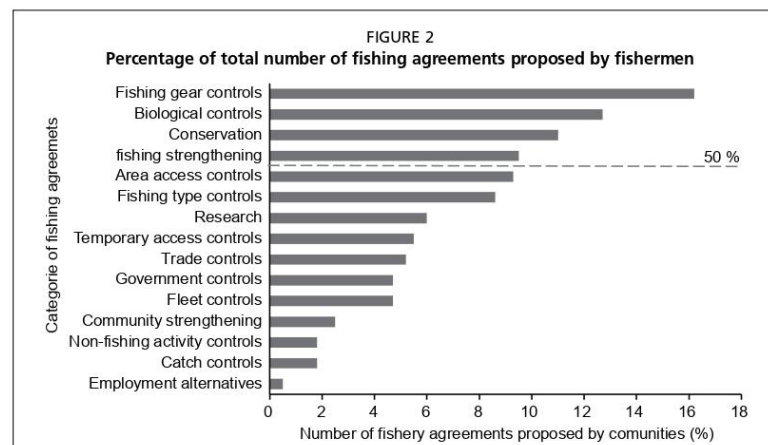


TABLE 4  
Similarity percentages SIMPER for the average of fishery measures proposed for each fishing agreement category

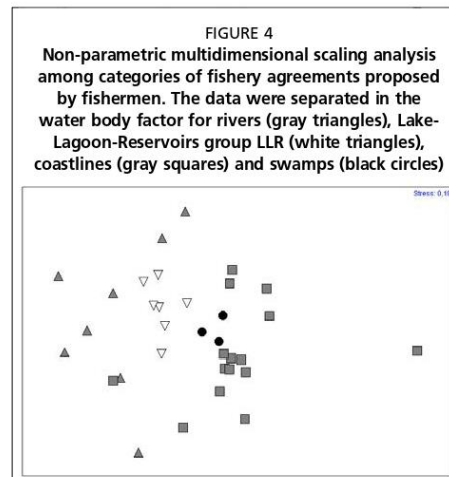
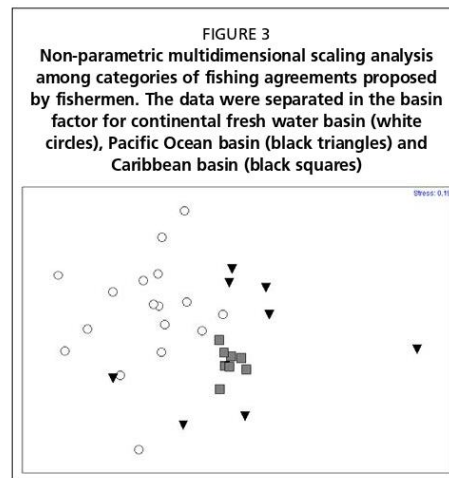
Fishing agreements categories	Basins			Water bodies				Type of fishing resources		Type of aquatic environment	
	Pacific	Caribbean	Continental	River	LUR	Coast	Swamp	Ornamental	Fishing-based consumption	Fresh waters	Marine waters
Biological controls	0.63	1.80	1.12	0.50	1.86	1.24	1.33	0.57	1.33	1.11	1.25
Fishing strengthening	1.38	1.80	0.29			1.65	1.67	0.00	1.22	0.39	1.63
Fishing gear controls	0.88	1.90	1.41	1.13	1.57	1.41	2.00	1.00	1.52	1.33	1.5
Area access controls	1.13	0.00	1.12	0.50	1.86	0.53	0.67	0.43	0.93	1.22	0.38
Conservation	0.50	1.30	0.82	0.25	1.00	0.76	3.00	0.00	1.00	0.78	0.81
Fishing type controls	0.50	0.90	0.94	0.75	1.14	0.65	1.33	0.86	0.78	0.94	0.63
Trade controls	0.50	0.80	0.35	0.50	0.14	0.65	0.67	0.57	0.48	0.39	0.63
Fleet controls	0.13	0.90	0.24	0.00	0.57	0.47	0.67	0.00	0.44	0.22	0.5
Government controls	0.75	0.70	0.24			0.65	1.33	0.00	0.56	0.28	0.63
Community strengthening	0.25	0.50				0.35	0.67				
Research	0.63	1.00	0.35	0.25	0.29			0.29	0.67	0.39	0.81
Temporal access controls			0.71	0.63	0.71	0.18	1.00	0.57	0.41	0.72	0.13
Catch controls				0.00	0.57						
Non-fishing activities controls				0.25	0.14	0.00	1.00				

For the basin and water bodies factors, at least two variables were differentiated; therefore not only a global R was calculated, but also particular R for each pair of data comparisons (Table 3). Fishery agreements proposed between coastlines (Caribbean and Pacific basins) showed partial overlapping with some differences ( $R=0.428$ ,  $P=0.001$ , ANOSIM). Fishery agreements proposed by Caribbean fishermen gave greater importance to fisheries categories such as biological controls, fishing gear controls, conservation issues and fleet controls, while Pacific Ocean fishermen gave priority to area access control.

A partial overlap in fisheries agreements for continental freshwater basins and both coastal fisheries was found ( $R=0.361$  and  $R=0.274$ ,  $P=0.001$ , ANOSIM). Area access control was important in fisheries agreements for continental freshwater basins and Pacific Ocean basin, but not for Caribbean fisheries. However, fishery categories related to fishing strengthening and government controls were low or absent in Pacific Ocean fishery agreements, but with higher relevance in continental freshwater basin fishery agreements.

On the other hand, similarity was found among fishery agreements in the Caribbean, while fishery agreements in the Pacific Ocean fell into two groups (Figure 3), whereby there were different contexts, interests and ways to face problems among small-scale fishery communities from Pacific and Caribbean basins. Fishery agreements proposed in continental freshwater basins were different from those for coastal fisheries, even though continental freshwater basin fishery agreements showed a scattered distribution (Figure 3), according with their own fishery measures. This indicated that some differences were identified in fisheries categories among freshwater fishery communities.

Categories of management proposed by fishermen for rivers and coastal areas were different ( $P=0.001$ , ANOSIM), and both groups showed a certain dispersion



pattern (Figure 4). Fisheries measures such as government control, fleet control, fisheries promotion, and community strengthening were not considered by river fishery agreements, but were considered in coastal agreements. Alternatively, coastal fishery agreements considered biological controls and conservation issues as very important, while river agreements considered the temporary access control as one of the most important fishery categories.

Fishery agreements in swamps and other freshwater systems, such as lakes, lagoons and dams (LLR) showed results in different ways. A difference was found between fishery agreements in swamps and LLR ( $R=0.627$ ,  $P=0.001$ ). Swamp fishermen preferred measures related to fishing gear controls and closures, together with measures regarding conservation issues and fishing promotion. In addition, these swamp fishery agreements considered fishery agreement categories like government controls and community strengthening, but these were not included in LLR fishery agreements. LLR agreements, however, gave priority to area access controls and catch controls.

LLR fishery agreements partially overlapped with coastal and river fishery agreements ( $R=0.325$  and  $R=0.297$ ,  $P=0.001$ ). Fishery agreements in these water bodies emphasized fishery measures associated with biological controls and area access controls, and gave less importance to trade controls. On the other hand, fishery agreements for swamps showed significant similarities in fisheries agreement categories such as area access controls, trade controls and catch controls, with river fishery agreements. They

also showed similarities with coastal agreements in fisheries categories such as biological controls, fishing gear controls and fishing promotion. Therefore fishery agreements for swamps, coastlines and river basins showed no differences ( $P>0.05$ , ANOSIM). Finally, analyses for fishery agreements among ethnic groups (mestizo, indigenous and afro-descendants) showed no differences in fisheries agreement categories (Table 3).

## DISCUSSION

### Fishery agreements in the context of participatory management

Fishery agreements constitute the first steps for consolidating participatory management of natural resources (Graham *et al.*, 2006; Pomeroy and Rivera-Guieb, 2006). In fact, fishery agreements are not difficult to accomplish, if there is trust between local communities and the management team (De Castro *et al.*, 2000). This may explain in part the wide range of variation in the number of categories identified in fishery

agreements and in the number of measures for fishery management proposed by fishermen within those agreements. However, the number of categories of fishery agreements, their method of implementation, and effectiveness depend on the particular conditions in which the small-scale fishery is developed (Haggan *et al.*, 2007). In the case of Colombian small-scale fisheries, three external factors may influence the definition of these fishery agreements.

The first external factor is related to the limited capacity of government to identify priorities and establish clear criteria for participatory management. This situation makes it difficult to obtain legal supporting for small-scale fisheries given that there are many particular conditions (Lam and Pauly, 2010), for example the poverty degree, community isolation, public order problems, etc. As a result, many small-scale fishery agreements could be starting with different points of view in accordance with the goals of those institutions that participated in or led these processes. It may appear that these fishery agreements were framed in responsible fishery agreements (Agudelo *et al.*, 2013; Fundación Ecosfera - AUNAP, 2013; Zuluaga and Franco-Jaramillo, 2013; Trujillo *et al.*, 2011; Niño, 2011; Trujillo and Trujillo, 2009), fishery management plans (Fundación Humedales - AUNAP, 2013; AUNAP, 2012a; AUNAP, 2012b; Barrero *et al.*, 2010), conservation agreements (Niño *et al.*, 2013; Delgado *et al.*, 2010; PNN SANQUIANGA, 2009), trade agreements (Red de Frio, 2012), sustainable fishery practices (Fundación Ecosfera - AUNAP, 2013; Fundación Humedales - AUNAP, 2012), and rules for fishery management (López *et al.*, 2006). Nevertheless the given name to each participatory management process by their stakeholders, included in any of above management concepts, it necessarily will not mean similar mechanisms neither way of implementation. The second external factor pertains to the different management perspectives that into the generation of fishery agreements. The participation of different stakeholders such as the government (e.g. Fisheries Authority, Environmental Authority) NGOs, research institutions, universities, and trade sector, among others, were not balanced in all 36 participatory process. Most small-scale fisheries management processes have been led by private institutions; NGOs in particular have played an important role in generating ideas and methods for developing participatory management approaches (McConney and Charles, 2008; Kurien, 2003) in situations where the government has limited capacity or is entirely absent. The goals of such processes were directed by these institutions depending upon their own view of various participatory management strategies, including Community Based Management -CBM- (Graham *et al.*, 2006; De Castro *et al.*, 2000), Co-Management - CM - (Hartoto *et al.*, 2009; Berkes, 2008; Armitage *et al.*, 2007), Co-Management Based-Fishery Community Management - CBFCM; Community-Based Resource Management - CBRM- (Haggan *et al.*, 2007; Pomeroy and Rivera-Guieb, 2006), Right Based Management -RBM- (WWF, 2013), among others (McConney and Charles, 2008; Armitage *et al.*, 2007; Haggan *et al.*, 2007; Pomeroy and Rivera-Guieb, 2006; De Castro *et al.*, 2000). Although all these approaches for management are based on community participation in decision making and management, the level of participation of other stakeholders, including the government, may differ.

As a result of the first two external factors, a third external factor emerges, referred to as the lack of alignment and prioritization of category agreements and methods of enforcement. It may be possible that small-scale fishery processes transversal to social and ecologic aspects directly related to communities, as well as the role of government were not completely taken into account (Armitage *et al.*, 2007; Pomeroy and Rivera-Guieb, 2006; Carlsson and Berkes, 2005; Charles, 2005). As such, the fishery agreements that included control of fishing gear, control of biologic parameters of target species in the fishery, resource and ecosystem conservation and fishery promotion reached together a 50% prioritization over 15 total kind of fishery agreements proposed by fishermen, which is considered important. On the other hand, other kind of agreements

related with area/temporary closures included the national fishery law (Law 13 of 1990) and the government participation had low priority by fishermen.

A low or limited presence of the government in fishery management proposals could be more in line with CBM, apparently for the historic absence of government for local decision-making. However switching from a model based on government management to a model with a wide range of local community autonomy may be not the best choice for small-scale fishery management in Colombia, even in protected areas such as National Parks, indigenous areas, or territories managed by afro-descendant groups.

Enforcement of participatory management in Colombia will require an improved level of community organization in order to gradually empower community members for managing fisheries resources, without overlapping with the responsibilities of national fishery authority as Pauly (2006) suggest. Different levels of co-responsibility related to fishery management participation in the fishery agreements proposed by the communities studied could be due to the following reasons: i) small-scale fishermen strongly believe that the resources from the sea or river are essentially inexhaustible, ii) the historical perception based on experience, and transmitted from father to son, about fishery resources are not disappearing, but they are moving to other areas or deeper waters, as an effect of increased effort on the fishery; iii) fishermen's livelihoods are generally supported by daily catches with limited middle- to long-term planning; and iv) fishermen tend to think that fisheries-related problems are mostly caused by external factors (e.g. environmental damage) and not by their own activities.

#### **Diversity of fishery agreements in Colombia**

Small-scale fisheries in Colombia integrates many different approaches. The country has two coast lines (Pacific and Caribbean) and numerous large and small river basins, all of them with different realities. Fishermen of coastal zones prioritized fishery agreements to increase their incomes and enhance organization, but these preferably provided for agents external to their communities and fishing areas. This preference is possibly promoted by the historical interaction with actors of the government and private sectors, given the multiple economic interests associated with coastal waters (tourism, oil companies, etc.). On the other hand, fishery agreements in river systems prioritized controls for fishery access with respect to area and time, recognizing in this case, compared with marine fishermen, that fisheries management also depends upon fishermen as local users. This may be related to the recognition of a drastic reduction in fishery resources by river fishermen (Gutierrez, 2010), showing that in just few circumstances the resources may sometimes be replaced by other species not previously considered as commercials (e.g. ornamental fishery).

Coastal fishermen appear to have a more optimistic vision than inland fishers, where reductions in traditional and commercial species in recent decades have been compensated by market adaptation introducing new species into fisheries (García *et al.*, 2007). New species introduced for trade partially balance the incomes lost to species that are no longer caught. Fishermen of the Pacific coast emphasized controls of area access, due to the conflicts small-scale fishermen have with industrial fleets in recent decades, especially with the shallow water shrimp fleet (Rueda *et al.*, 2006) and the deep sea shrimp fleet (Díaz *et al.*, 2011), that after a five-year process finally allowed the creation of an Exclusive Small-scale Fisheries Zone (ESSFZ) and Special Zone for Fisheries Management (EZFM) in northern Pacific coasts of Colombian pacific (AUNAP, 2013). These areas point out a new legal frame to develop participatory management schemes in the country.

Fishermen that target fisheries for consumption showed a greater need for fisheries regulations and investment by fisheries authorities than fishermen targeting ornamental fishes. These results can be interpreted in different ways; the ornamental

fishing is usually conducted in areas with low accessibility (Zuluaga and Franco-Jaramillo, 2013) that promotes geographical isolation for fishermen (López *et al.*, 2012). On the other hand, other phases of the fishery such as resource gathering and national and international trade require participatory management of fishery resources to be enforced at multiple geographic scales (Berkes, 2008). Ornamental fishermen and consumption-based fishermen perceive that the legal and illegal mining (e.g. gold mining in rivers) poses one of the highest risks to small-scale fisheries, including both the ecosystem and fishermen themselves. As such, ornamental fishermen are addressed to implement management measures of the kind they are able to enforce, while fishermen targeting consumption-based resources feel safe to enforce several measures, even when some of them cannot be enforced in the appropriate manner. There are many reasons for an inappropriate enforcement of management measures that have been proposed by the fishermen, however appear that a community weak structure usually is related with low implementation or breach of fishery measures.

Agreements in river fisheries prioritized temporary controls such as closures, as fishermen recognized that successful catches are directly linked to the rainy season and river water levels (Hernandez, 2004; Poveda, 2004). As such, poor enforcement of fishery measures during peak fish vulnerability could ultimately have negative effects on the fishery. It is also recognized that some species are highly migratory, and fishermen take advantage of this condition by increasing fishing effort during migration. This is the case of the striped catfish *Pseudoplatystoma magdaleniatum*, which is overfished during its spawning season while performing its upstream migration known locally as “subienda” (Hernandez, 2004).

Fishery agreements in lakes, lagoons and reservoirs (LLR) or mixed waters like estuaries and swamps were different from other types of agreements. In this case, area and capture controls were, as these measures are easier to enforce in LLR than estuaries during the year. Fishery agreements in estuaries prioritized similar actions to those in rivers and coastlines. Fishermen here find the government as an important stakeholder for setting regulation measures, but also for research and fishing strengthening. They also think other activities besides fisheries should be regulated. Fishermen recognize estuaries and swamps as complex areas with diverse interests that not only include fisheries (Carlsson and Berkes, 2005), and actually increase uncertainty and fragility to fisheries (De Castro *et al.*, 2000).

Colombia is a multiethnic country comprised of indigenous and afro-descendant communities that have their own territories protected by national law (DANE, 2007). However, this analysis showed that fishermen proposed similar fishery agreements, regardless of their ethnicity. This suggests that community perception of fisheries management moves beyond the similar view point, but that each community can propose culturally-appropriate mechanisms to enforce fishery measures under a fishery agreement within their territory (Haggan *et al.*, 2007). Results showed similar factors affecting fishery communities regardless of ethnic distinction, such as overfishing (Gutierrez, 2010); limited regulatory capacity of the government (Haggan *et al.*, 2007); centralized management decision-making (Armitage *et al.*, 2007); poor link between scientific knowledge and traditional knowledge (Berkes *et al.*, 1991; Nielsen *et al.*, 2004); poor empowerment of communities for fishery management (Osorio and Betancur, 2007); diffuse mechanisms for sanctions (McGrath *et al.*, 2008); global changes in supplies, demands, value and use of fishery resources (Pomeroy, 1995) and poverty and alternatives loss (Charles, 2005).

Although similarities among fishery agreements were found among agreement categories in different regions and for different water bodies, there were different fishery measures within an area or water body. Such differences may be explained by the high variability in community organization, regardless of ethnicity. A relationship between the level of community organization and the level of fishery measures

proposed was observed, implying that the more organized communities are, the more structured fishery categories people proposed.

#### **Toward a fishery participatory management enforcement**

The effective distribution of power for fisheries management should be understood as a result and not as starting point of a participatory management scheme, so that legal and formal aspects of enforcement do not displace functional aspects (Carlsson and Berkes, 2005). It is necessary to recognize that in Colombia, participatory management of natural resources is a dynamic process (McConney and Charles, 2008; Pomeroy and Rivera-Guieb, 2006) that varies with social and ecological contexts (Crowder *et al.*, 2008; Armitage *et al.*, 2007).

The definition of different fishery categories according to the conditions of small-scale fisheries make it evident that general governmental criteria to validate and allow management in all different areas such as river basins, coastlines, and between consumption-based and ornamental fisheries are needed. It is also necessary to adjust management criteria according to the type of community (which are quite diverse in Colombia) to optimize the effect of fishery measures (Haggan *et al.*, 2007) as well as the role of fishermen in fishery agreements (Kurien, 2003). If small-scale fisheries are considered in an integrated management system, only then can they achieve sustainability and resilience (Charles, 2005).

Success in the enforcement of participatory management in developing countries with limited government capacity need to develop new legal and administrative baselines and institutional agreements to enhance current political, social, cultural and economic situations (Pomeroy, 1995). In order to achieve this goal, government, communities and other relevant stakeholders need to work together to ensure that data is gathered with confidence and transparency, and all stakeholders develop a sense of ownership with respect to fishery resources (Lam and Pauly, 2010; Fundación Ecosfera, 2009), as well as assume responsibilities in management process (Polanco and Rodríguez, 2013). This implies that the anthropocentric approach, whereby human beings are considered unidirectional receiver of goods and ecosystem services, must be replaced by an approach where the man is responsible for ensuring that ecosystems maintain health and integrity and thus provide requirements which we depend. Such an approach should be reflected by any participatory ecosystem-based management scheme (Bundy *et al.*, 2008).

At a national level, and more so in international conventions (WECAFC, IATTC, South Pacific RFMO, SPPC) and fishery organizations (OSPESCA) with which Colombia is involved, it is evident that participatory management is important for achieving the goals of fisheries management. Colombia has many challenges to face, due to water pollution, waste water, sedimentation, aquatic ecosystem fragmentation and deforestation (Steer *et al.*, 1997), as well as the huge challenge to regulate small-scale fisheries activities such as fishing effort, low selectivity fishing gears and methods, and low compliance of fishery measures such as closures.

For participatory management of fisheries in lakes, lagoons and reservoirs (LLR), it is important to take into account the regulation of fishing efforts at the mouths of rivers where fishes arrive or migrate to spawn, as well as the selectivity of fishing gear (INCODER, 2012; Trujillo and Trujillo, 2009). Freshwater fisheries also need to attend to issues regarding exotic invasive species establishment in rivers and dams where small-scale fisheries operate. These situations should be viewed within a participatory management framework, as these species already make up part of the capture for local communities, despite the institutional limbo between national environmental and fisheries authorities.

In conclusion, the implementation of participatory management in Colombia should include a clear government-based approach to: 1) incorporate the diversity

of ways in which small-scale fisheries are developed, 2) make an effort to establish alternative and complementary policies, in addition to conventional fisheries policies, to apply participatory management according to the characteristics of fishermen in different types of communities, 3) develop mechanisms to apply adaptive participatory management, 4) consider the cost of implementation of participatory management for both government and communities, and 5) establish legal frameworks that guarantee long-term implementation of participatory management.

This analysis demonstrates the need to review and align government and community enforcement and the roles of stakeholders and NGOs together with effective and reliable data (based on traditional and scientific knowledge) to promote egalitarian negotiations adjusted to the realities of communities and fisheries, and achieve effective participatory management schemes. The comprehensive fishery dynamics analysis, allocation of rights to public lands for the purposes of fishing in rivers, lagoons, lakes, estuaries, and the sea, and the important role of socio-cultural patterns and lifestyles of fishermen are challenges facing government institutions such as the maritime, environmental, ethnic and fisheries authorities, among others. From a community perspective, the effective establishment of participatory management schemes will depend on the community's level of organization. Furthermore, their commitment to assuming co-responsibility (with rights and duties) under this system will be the real platform for the establishment of long-term participatory management.

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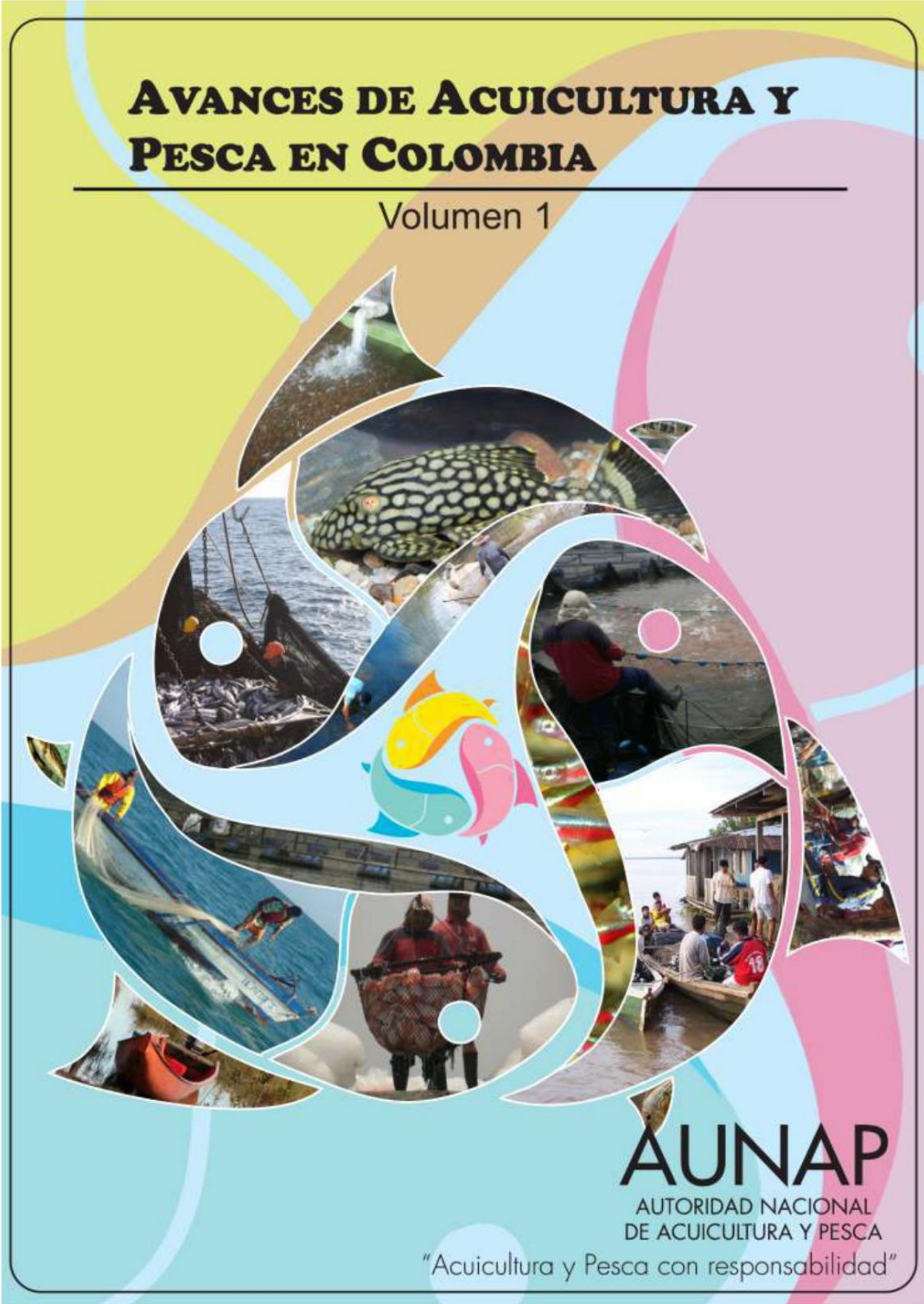
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## EVIDENCIANDO LA NECESIDAD DE GESTIÓN EN LA PESCA ARTESANAL DE COLOMBIA: EL CASO DE LA PESCA WAYUU EN LA GUAJIRA, CARIBE COLOMBIANO

John G. Ramírez<sup>12</sup>

En Colombia, la pesca artesanal es comúnmente percibida como un bien cultural, y en ocasiones turístico de muchas ciudades y regiones del país, pero no suele considerarse como parte importante de la sociedad. Según estimaciones del número de pescadores que habitan las costas del Océano Pacífico y Mar Caribe colombiano (Beltrán y Villaneda, 2000), una población equivalente a la encontrada en la ciudad de Tunja, con alrededor de 160.000 personas, depende de la pesca artesanal marina. En un contexto netamente económico, el sector de la pesca y la acuicultura representa alrededor del 0.5% del PIB nacional, y del total de éste la pesca artesanal solo representa el 20% de la producción (Robles, 2008; Puentes *et al.*, 2014;). Por consiguiente, bajo una perspectiva económica se estaría subestimando la importancia social del subsector artesanal.

La pesca Wayuu en la media Guajira ha representado entre el 24% y 31% de los desembarques respecto a la captura total del Caribe continental colombiano, siendo la más importante de la región (CCI, 2010, 2006; AUNAP, 2014). Estas destacadas capturas están asociadas a condiciones de alta productividad en una zona que representa solo el 5% del litoral Caribe (Páramo *et al.*, 2003; Manjarrés, 2004; Andrade y Barton, 2005; Criales-Hernández *et al.*, 2006; Sarmiento-Devia *et al.*, 2013). A pesar de su importancia, la población Wayuu, incluyendo a los pescadores, que en su mayoría habitan zonas rurales, poseen un índice de necesidades básicas insatisfechas entre el 85.45% y 98.28% (DANE, 2005), lo que implica que esta población indígena vive en condición de miseria.

La pesca en la media Guajira abarca todo el litoral desde el corregimiento de Camarones, Municipio de Riohacha, hasta el sector de Santa Rosa, en el Municipio de Manaure (Fig. 1). Existen decenas de comunidades pesqueras apostadas en la línea de costa y esto implica que los desembarques de pesca se hacen por múltiples puntos en playa. Un punto crítico es el desconocimiento por parte del sistema de monitoreo oficial (SEPEC, POPC) de la pesca desembarcada en la zona rural, debido a que el monitoreo de desembarques que efectúa la agencia pesquera del Gobierno solo incluye los puntos urbanos. Para resolver este asunto, desde el año 2006 se lleva a cabo un monitoreo participativo en cada una de las rancherías, contando a la fecha con más de 90.000 registros de desembarque (Fundación Ecosfera, 2009).



Figura 1. Embarcación de pesca Wayuu, Media Guajira. Fuente: John G. Ramírez<sup>o</sup>

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Por otro lado, la pesca artesanal involucra al menos 11 artes de pesca y tres tipos de embarcaciones con diferente autonomía. Como consecuencia, la pesca Wayuu captura al menos 167 especies distintas (Puentes *et al.*, 2012), siendo esto un reflejo de la diversidad de esta pesquería. Determinar el estado de las poblaciones de todas las especies capturadas es poco probable, y menos aún con la limitada información disponible, por lo cual es razonable priorizar el estudio de aquellas que pudieran representar los intereses de las pesquerías (Hilborn, 2011). De este modo, el pargo chino *Lutjanus synagris* es la especie íctica más valorada, siendo un objetivo principal de captura de la pesquería Wayuu. Adicionalmente, la boca colorada *Haemulon plumierii* cobra mucha importancia desde un punto de vista de seguridad alimentaria y cumplimiento de compromisos culturales en el interior de la comunidad Wayuu. Además esta especie reportó las mayores capturas con respecto a cualquier otra especie dentro de la pesquería (Ramírez *et al.*, en prep).

Ramírez *et al.* (*En prep.*) determinaron que el pargo chino ha duplicado su estado de sobreexplotación entre el año 2006 y 2014. Por su parte, la boca colorada presentó un estatus de aún mayor sobreexplotación, sobrepasando en tres veces el nivel de pesca sostenible. Adicionalmente, las capturas disminuyeron hasta cuatro veces entre los primeros y últimos años del estudio. Lo anterior pareciera exagerado bajo la percepción de que la pesca artesanal no genera demasiado esfuerzo sobre los recursos naturales; sin embargo se evidencia una sobreexplotación desproporcionada por una de las flotas de la pesquería artesanal, la cual está llevando al colapso los recursos pesqueros, y a su vez a una aún peor situación a los pescadores.

Esta flota, denominada localmente como “lanceo”, la cual captura tanto por encierro como enmalle en aguas someras, pesca sobre distintos tipos de dispositivos agregadores de peces (DAP). El lanceo es una flota “no natural” y tampoco tradicional, que surgió por la interacción de múltiples factores asociados a la ausencia de gestión pesquera, condiciones ambientales y dinámica social. Aprovechando que la pesca en la media Guajira es somera, muchas embarcaciones que antes pescaban con redes de enmalle de fondo “transparente” o con redes a la deriva “caritero” aumentaron la longitud de las redes multifilamento a más del doble de altura. Ayudada por la experiencia de los buzos locales, así como de motores fuera de borda que fueron instalados en los botes tradicionales de madera, esta flota captura especies demersales de forma intensa sobre los DAP (Ej: llantas de camión, carrocerías de vehículos, entre otros).

La flota de lanceo ya se encontraba en desarrollo en el año 2000 (Manjarrés, 2004), mientras que en la actualidad domina los desembarques totales por encima de todos los artes tradicionales (Estadística de AUNAP y participativa Wayuu). Hacer la transición de una embarcación y arte tradicional, a esta nueva forma de pesca, no ha sido sencillo. La pesca de lanceo requiere una alta inversión, que suele estar fuera del alcance de los pescadores Wayuu. Lo anterior implicó inversiones de comerciantes locales, buscando alta rentabilidad pero con poca consideración de la importancia social de la pesca. Esto motivó a su vez que muchos pescadores provenientes de otros Departamentos del Caribe, donde este arte de pesca es usado para la captura de especies pelágicas, iniciaran una pesca demersal en la Guajira. De esta manera, no solo cambió la forma de pescar, sino que nuevos actores, y con nuevos intereses se involucraron.

A pesar de que la red de lanceo tiene una alta capacidad de captura, sigue siendo operada en una embarcación con eslora menor a 8 m, de madera o fibra de vidrio. Por lo tanto, esta flota de pesca, exhibiendo mucha más capacidad y efectividad por su método, se camufla como flota artesanal, pero con efectos mucho más devastadores. La red transparente, como arte más extensamente usado en la media Guajira, alcanza una captura anual promedio por embarcación de 822 kg, mientras que una embarcación perteneciente a la flota de lanceo, en promedio supera los 7500 kg al año. Cada una de estas flotas, tiene alrededor de 90 embarcaciones de pesca activas cada año. Por lo tanto, la fracción de los recursos pesqueros que remueve la flota de lanceo, es mucho mayor a la que es pescada por la flota que emplea la red transparente, como el método de pesca tradicional más usado.





Además de un gran poder de remoción de biomasa, el lanceo captura en su mayoría ejemplares de tallas pequeñas, por lo que este tipo de captura supone una sobrepesca de crecimiento que a su vez genera la sobreexplotación de toda la población (Beverton, 1963; Walters, 1992; Hilborn and Saari *et al.*, 2014; Froese *et al.*, 2015). En el caso del pargo chino, que recibe aportes de ejemplares de distintas tallas y de otras áreas por fuera de la zona de pesca de lanceo (Manjarrés, 2004; Rodríguez y Páramo, 2012), un colapso generalizado de la población se podría estar dando. Esto implica que el estado actual de aprovechamiento podría estar reflejando una drástica disminución de toda la población, que coincide con una fuerte reducción de la captura de ejemplares grandes en zonas más profundas y/o alejadas. La boca colorada es una especie que está asociada principalmente a la plataforma somera, y tanto juveniles como adultos poseen poca movilidad (Appeldoorn *et al.*, 2009; Villegas-Hernández *et al.*, 2014). Por lo tanto, la sobreexplotación trae aún peores consecuencias. Esto coincide con drásticas reducciones de la captura, incluso para la misma flota de lanceo.

A nivel social, las implicaciones son igualmente delicadas. La sobreexplotación de no solo estas dos especies, sino de todos los demás recursos que se encuentran bajo la misma presión, está conllevando la reducción de la captura de la inmensa flota artesanal tradicional en zonas rurales. Por lo tanto, la seguridad alimentaria y el cumplimiento de los compromisos sociales de miles de Wayuu se están viendo afectados. Esto es aún más preocupante si se considera que La Guajira exhibe los más altos índices de mortalidad infantil por desnutrición del país (Lissbrant, 2015). A su vez, algunos pescadores Wayuu de zonas rurales, viendo reducidas sus capturas, están buscando apoyo externo para introducir más embarcaciones para hacer pesca de lanceo. En consecuencia, se está causando más presión sobre los recursos y además se está minando el concepto social de la pesca artesanal, que en muchos casos, es el único sustento de estas comunidades.

Todo lo anterior, necesariamente conduce a reflexionar sobre la gestión que el Gobierno y sus distintas instituciones está llevando a cabo. A nivel de seguimiento pesquero, es claro que no contar con un monitoreo en zonas rurales ha producido una imagen de la pesca bastante distorsionada. A la vez, bajo la apariencia de buenas capturas registradas por el monitoreo oficial en zonas urbanas, se esconde un grave problema pesquero y social en zonas rurales. Las dificultades de la pesca artesanal Wayuu han superado estrictamente las consideraciones de una pesca sostenible y ahora también enfrentan problemas de índole humanitario. Lamentablemente, los entes territoriales locales tampoco han tenido en cuenta esta situación, mientras que sus limitadas intervenciones, están generando aún más agudización de los problemas mencionados. Por todo lo anteriormente descrito, la gestión de la pesca Wayuu y el control de la flota de lanceo no da espera.

Finalmente, varias reflexiones sobre la gestión de pesquerías artesanales en Colombia emergen de la situación de la pesca Wayuu. Más problemático aún que tener pesquerías bajo el sello de limitadas en datos (“data poor - fisheries” - Salas *et al.*, 2007; Chrysafi y Kuparinen, 2015), es no considerar adecuadamente el alcance real de los datos disponibles. Esto conlleva a grandes omisiones e interpretaciones erróneas de la realidad. ¿Qué está sucediendo con las pesquerías artesanales de nuestro país, sin que apenas lo sepamos? Una visión de la pesca (sobre todo artesanal) en función de lo que produce, más que en los miles de personas que sustenta, está causando daños irreparables sobre los recursos y la gente que depende de ellos. El caso de la pesca artesanal Wayuu de la media Guajira pone de manifiesto que es importante establecer una gestión efectiva de la pesca artesanal en el país. Por lo tanto, una débil o escasa gestión podrían conllevar al detrimento significativo de los recursos pesqueros, ecosistemas asociados a ellos, así como agudizar la situación social y económica de los pescadores. La implementación de un esquema de gestión y manejo participativo, la consideración científica de los datos disponibles, junto con un adecuado alcance a los resultados que se obtengan con dicha información, son urgentes (Ramírez y Puentes, 2015; Saavedra-Díaz *et al.*, 2015). Esto implica que cada pesquería artesanal sea adecuadamente dimensionada tanto por el gobierno, como por los actores estratégicos involucrados, incluyendo a los pescadores.

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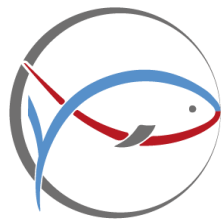
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# Fisheries Bioeconomic Modelling

