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Evidence of difference in landings and discards patterns in the English Channel and North Sea Rajidae complex fishery

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Abstract :

Bycatch species such as skates and rays are for most of them not subject to analytical stock assessment. However, their life history characteristics increased their vulnerability to fisheries. In the English Channel and North Sea area, the three main landed Rajidae species are Raja clavata, Raja brachyura and Raja montagui. The current management measure is a global TAC, common for all Rajidae species. Data to process analytical stock assessment are not available for these species, particularly discards data. A Bayesian multispecies biomass production model, following separately the landings and discards was applied to these stocks. This model provided proxies of reference points (MSY and BMSY) per species. All stocks were depleted in 1990 and are now rebuilding. However, rebuilding speeds are different within the complex, R. clavata being the fastest and R. brachyura the slowest. Furthermore, the proportion of the discards and landings to biomass differ between species, highlighting species specific fishing strategies. Differences in vulnerabilities within the Rajidae complex might be caused by the variability of life history parameters between species as well as landings and discards pattern differences. This second factor, usually not considered for data limited stock assessment, is particularly relevant for highly discarded chondrichthyans species, and might be considered when choosing new methodology to asses Rajidae stocks.

Keywords : Retention patterns, Vulnerability, Discards, Rajidae, Multi-species production model

1. Introduction

Worldwide, 34% of assessed fish stocks are overfished (FAO, 2020, Hilborn et al. 2020). In response, the demand for adequate management measures has been growing. Currently, quotas in a variety of forms (species specific, complex, multispecific) are widely used to regulate fishing pressure (Karagiannakos 1996, Marchal et al. 2016, Newell et al. 2005), with increased requirements for stock assessment models (Carruthers et al., 2014).

Since analytical stock assessments have primarily been implemented for commercially valuable species, until recently, little attention has been paid to bycatch species such as Rajidae, resulting in a relatively low data collection effort (Stevens et al., 2000). However, Rajidae are particularly sensitive to fishing, due to their specific life history traits (Dulvy et al., 2000). Most of these species are characterized by long life expectancy and low fecundity, which reduces their resilience to fisheries (Dulvy et al., 2014, Dulvy & Reynolds, 2002). Furthermore, being top predators, Rajidae play an active role in the ecosystem's top down regulation. All of these factors underline the fact that a decline of Rajidae is considered as a potential threat to the whole ecosystem (Stevens et al., 2000).

In the English Channel and North Sea (ECNS) area, 3a, 4 and 7d ICES (International Council for the Exploration of the Sea) statistical rectangles, Rajidae are mainly taken as a bycatch in flatfish fisheries. Most of these by-catches are taken by trawlers and to a lesser extents gillnetters, from France, the United Kingdom, Netherlands and Belgium. The mandatory reporting of landings by species for Rajidae in the ECNS only started in 2009. Before 2009, all Rajidae landings were binned into a single category. Although misidentification might occurred the reports from 2009 to 2018 are supposed to be consistent with no clear improvement or deterioration of the quality of identification. In addition, the absence of otoliths and the complexity of vertebrae age-reading has hindered the provision of age-structured proxies for relative abundance. Hence, with little data available to support analytical stock assessments, Rajidae have been considered Data Limited Stocks (DLS) and assessed with trend-based analysis methods by the ICES (ICES, 2019a).

Worldwide, about 80% of fish stocks are not analytically assessed, or not assessed at all, principally due to data limitations (Costello et al., 2012). Consequently, models have been developed to assist stock assessments in data-limited situations, with a majority of them based on surplus-production models, such as the Schaefer (1954) or Pella Tomlinson (1969) models (Pedersen & Berg 2017, Martell & Froese, 2013, Wetzel & Punt, 2015). Surplus-production models require less information compared to age- or size-structured stock assessment models. Still, reliable time series of catch data and one or more biomass indices are usually needed. The catch data are used to estimate the removals from the stock; those fish that die from fishing. This type of model provides proxies of reference points such as the biomass (B_{MSY}) or the fishing mortality (F_{MSY}) at the maximum sustainable yield (MSY) that are currently used to provide stock management advices. However, they might have high sensitivity to inputs (Geng et al., 2020) and not explicitly take into account age structure and the existing delays for elasmobranch between reproduction and recruitment (Bonfil, 2005). Surplus production models have already been applied to assess several sharks and rays species, in the USA and Australia (Bonfil, 2005, Cortès et al., 2002) and might be used to assess some of the ICES data limited stocks.

The landings data of Rajidae underestimate the total catches, and thus the removals. They are mainly taken as bycatch species, their TAC is usually reached before the main target species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*) TACs, and a substantial fraction of Rajidae are discarded. In the case of Rajidae stocks in the ECNS, discards data come from observer programs that were first dedicated to flatfish, in the area (ICES, 2003). For that reason, Rajidae discards data are sparse.

Discarding in demersal fisheries in the ECNS occurs for several reasons. First fish may be smaller than the minimum size restrictions set within the fishery (Feekings et al., 2012). In 2020, the Rajidae minimum landing size varied between countries with, a total length of 45cm in France, 50cm in

Belgium and 55cm in Netherlands and a total width of 40cm in the United Kingdom. These four countries fleets occur constantly in the area and represent the essential of Rajidae catches. These country specific minimum landing sizes were already implemented at the end of the time serie used. In addition to size restrictions fish may be discarded when quota limit landings of certain species while fishing takes place for other species (Batsleer et al., 2015). In this context, Rajidae are discarded to avoid a choke species effect on flatfish, but all Rajidae species, within the complex, might not be affected in the same way by this behaviour. This creates a mutual dependency between landings and discards that makes it essential to estimate their relation dynamics and causality (Rochet & Trenkel, 2005, James et al., 2016). A deep understanding of these elements might offer key elements to estimate catch only model suitability and understand factors that drove retention patterns.

The objectives of this study, based on the three main ECNS Rajidae species: *R. clavata*, *R. brachyura* and *R. montagui*, were, (1) to describe discards and landings trends among species, (2) to assess, based on a complex state space Bayesian model, the impact of discards trends on species stock status and biomass trajectories, and provide proxy reference points and (3) to identify potential reasons of increase in discard rates for some species while not for others.

2. Material and Method

2.1. Data

The surplus-production model used to estimate stock status for the three species requires biomass indices and catch data. In the ECNS Rajidae complex, these data were not directly available in a usable form.

2.1.1. Abundance indices

Biomass indices for the *Rajidae* stocks under study were estimated from the 4 main bottom trawl surveys that are conducted on an annual base in the ECNS: the North Sea International Bottom Trawl Survey (NS-IBTS) (quarters 1 and 3), the Channel Groundfish Survey (CGFS) (quarters 3), and the Beam Trawl Survey (BTS) (quarter 3). The 4 surveys were combined into a single biomass index. This was done to reduce the potential bias related to differences in spatial overlap of the surveys with the spatial distribution of the stock (Kraak et al., 2009) which is likely the case for the skate populations under study given theirs heterogeneous spatial distribution in the ECNS (Maxwell *et al.* 2009; Daan *et* al., 2004; Heessen *et al*., 2015).

Survey data as provided in the exchange format through the ICES DATRAS portal was used to derive a cpue (kg/km²) estimate at each sampling station. The swept area of each haul was estimated based on the available information on haul duration, fishing speed, shoot and haul locations, and information on technical gear characteristics and gear deployment (e.g. beam width for the BTS, and doorspread for the NS-IBTS and CGFS). To reduce variation caused by the use of different gear types between research vessels within a single survey, as is the case for the NS-IBTS, the analysis was restricted to the predominant gear type, being the Grand Ouverture Verticale (GOV). An overview of the surveys used for the index calculation is given in Table 1.

Table 1 Overview of the scientific surveys used in the index calculation of blonde, thornback and spotted ray in the ECNS.

	Period	Quarter	Gear	Countries	Min/ median/ max nr stations by year
NS-IBTS Q1	1990 – 2019	1	GOV	ENG, SCO, DEK, SWE, GFR, NED, FRA, NOR	328 / 378 / 428
NS-IBTS Q3	1991 – 2019	3	GOV	ENG, SCO, DEK, SWE, GFR, NED, FRA, NOR	0 /326 / 380
BTS	1990 – 2019	3	Beam trawl	BEL, ENG, GFR, NED	177 / 312 / 389
CGFS	1990 – 2019	3	GOV	FRA	59 / 87 / 106

Numbers-at-length were converted into weights by applying an allometric length-weight relationship with parameters taken from the ICES Working Group on Elasmobranch Fish (WGEF ICES, 2019c). Subsequently, the individual weights of species (≥ 50 cm for vulnerable biomass, and ≥ 0 cm for total biomass) were aggregated at the haul level to derive a cpue measurement for each haul. These cpue values were averaged by survey, year, and ICES statistical rectangle. This approach ignores potential variability related to vessels effects, however, this variability is assumed to have a minor effect since each survey is subjected to a predefined sampling design including guidelines on the fishing method, gear configuration and catch processing (ICES, 2019b; WGEF ICES, 2019c). The average cpue estimates by survey, year and ICES statistical rectangle were used to estimate differences in catchability between the surveys relative to each other (NS-IBTS vs BTS, and CGFS vs BTS) taking into account spatio-temporal differences in abundance. Catchability differences were estimated using observations that overlap in space (similar ICES statistical rectangle) and time (similar year) using logistic regression according to the following formula, $weight_{IBTS/CGFS}/(weight_{IBTS/CGFS} + weight_{BTS}) \sim \beta_0$, with the intercept ($\beta_0$) being the proportion of fish caught by the IBTS/CGFS compared to the BTS survey. Differences in catchability between surveys are thus assumed to be constant in space and time. The estimated model coefficients were used to rescale the NS-IBTS and CGFS cpue estimates to the BTS cpue, and averaged over the different surveys by ICES statistical rectangle and year. Finally, each cpue index was raised to the surface of the corresponding ICES statistical rectangle, and aggregated by year, to derive an absolute, annual index on the scale of the ECNS. A more detailed workflow is available in the supplementary materials (S1).

To test the validity of combining the NS-IBTS Q1 survey with the Q3 surveys (NS-IBTS, BTS, CGFS), the procedure above was applied to the data excluding the NS-IBTS Q1 and Q3 survey, respectively. These leave-one-out indices were compared with the combined tuning indices, using both the Q1 and Q3 NS-IBTS data. For the latter indices, confidence intervals were constructed using a non-parametric bootstrap analysis. A stratified resampling scheme according the year, the ICES statistical rectangle and the survey, was used to generate 1000 datasets to which the approach as outlined above was applied. The resulting test statistics were used to calculate 95% confidence intervals.

2.1.2. Removals

Landings data for the three species *R. clavata*, *R. brachyura* and *R. montagui* were extracted from the WGEF landings table over the period 2009-2018 (ICES 2019a). Landings data for the Rajidae complex for years 1990-2009 were extracted from the Historical Nominal Catches 1950-2010 (Eurostat/ICES, 2011). The Rajidae complex is composed of seven species including six commercial species, *Leucoraja naevus*, *Raja microocellata*, *Raja undulata, Raja clavata, Raja brachyura* and *Raja montagui.* Species-specific landing ratios were calculated for each commercial species of the complex using the species-specific data from 2009 to 2018. These yearly species specific ratios were averaged by species over the period and finally aggregated. The combined landings of *L. naevus*, *R. microocellata* and *R. undulata* represented a mean percentage of 9.8 % [9.6-9.9] of the landings from 2009 to 2018. This percentage was deducted from the landings in 1990 - 2008 to obtain an estimate of *R. clavata*, *R. brachyura* and *R. montagui* landings during this period.

Discards data were extracted from the WGEF discards table (ICES, 2019a), with information going back to 2009. Reported discards increased by a factor of 3 from 2010 to 2018 (766 to 2199 tonnes).. A multiple regression was applied to test the correlation between fishing effort by fleet and discards (S3), with effort data extracted from WGMixFish (ICES, 2017). Using the new discards data obtained from the regression, a ratio between landings and discards for the 2009-2018 time period was calculated. This ratio was then applied to the landings in order to get an estimate of the discards for the previous period (1990-2008). The hypothesis was made that discards did not stay stable for the entire time period with no change in fishing effort resulting in a clear increase in discard quantity after management measures were implemented. To this end, the ratio between landings and discards was reduced by a factor of 2 during the 1990-2008 time period.

Not all discards are part of the removals in fish stock assessments: the survival of discards needs to be taken into account when integrating them into the removals in stock assessment models (Davis, 2002, Benoît et al., 2013). In the ECNS, Rajidae discard survival ratios have been estimated to be between 0.53 and 0.55 in demersal and pulse trawl fisheries, with a focus on *R. clavata* for pulse trawls (Enever et al., 2009, Schram & Molenaar, 2018). To correctly reflect the actual amount of removals that the discards represent, the discard estimates were divided by 2, before being entered in the surplus-production assessment model.

2.2. Assessment model

2.2.1. Multispecies State Space Bayesian model formulation

A multispecies approach has been developed by Marandel et al., 2019 for the Bay of Biscay Rajidae complex. This approach made it possible to integrate multispecific data from before species specific catches report (2009). It extended the catch time series of 20 years and gave the opportunity to assess the relative behaviour of the different species within the Rajidae complex.

Based on Marandel's formulation, a multispecies model was developed to track discards and landings variations. Catches were split between landings and discards in the population dynamic simulation (1),

$$(1)\ L_{t,e} \sim bin(R_{t,e}, L_{OBS}),$$

$$D_{t,e} \sim bin(R_{t,e}, D_{OBS})$$

$$C_{t,e} = L_{t,e} + D_{t,e},$$

with $R_{t,e}$, the biomass ratio by species, e, at time t; $L_{t,e}$, the landings by species, e, at time t; L_{OBS}, the observed landings used as inputs, $D_{t,e}$, the discards by species, e, at time t, and $C_{t,e}$, the total catch by species, e, at time t. Landings and discards differentiation were then added in the observation units to determine divergence in ratio between biomass dynamics and landings as well as discards (2).

$$(2)\ 1/\tau_{Le} \sim gamma(400,1),$$

$$1/\tau_{De}(400,1),$$

$$P_{Lt,e} \sim N(R_{t,e}, 1/\tau_{Le}),$$

$$P_{Dt,e} \sim N(R_{t,e}, 1/\tau_{Le})$$

τ_{Le} and τ_{De} are the observation error terms for landings and discards by species, e, $P_{Lt,e}$ and $P_{Dt,e}$ the observed proportions per species, e, in landings and discards at time, t.
The MSY and B_{MSY} were estimated as (3)

$$(3)\ MSY = r.K/4$$

$$B_{MSY} = K/2$$

2.2.2. Priors settings and scenarios

Priors were set using Marandel's et al. (2019) approach. Four models were built, the same priors were used for all of them, for r (intrinsic growth rate), K (carrying capacity), q (catchability), τ (observation error) and σ (process error) (Table 2).

Table 2 Common priors r intrinsic growth rate, K carrying capacity, σ process error, τ observation error, q catchability.

Species	r~Beta mode, SD	Landings	Discards	K~Uniform min, max	$1/\sigma^2$~Gamma mode, SD	$1/\tau^2$~Gamma mode, SD	q~Uniform min, max
Raja clavata	0.105, 0.1	2009-2018	2009-2018 50% discards	8 000, 150 000	400,1	400,1	0.0001 ,0.0004
Raja brachyura	0.091, 0.1			1 000, 100 000		400,1	

Raja montagui	0.114, 0.1			1 000, 150 000		400,1	

Four scenarios hypothesized a stock substantially depleted (Table 3), with the initial biomass (*Y0*) from 1990 being 0.3 of the pristine biomass (M1, M3, M4). This assumption was based on general European post World War fisheries dynamics (Davidson et al., 2016, Letaconnoux, 1948). However, the hypothesis of a slightly less depleted stock with a mean relative biomass (*Y0*) from 1990 being 0.5 of the pristine biomass, was tested (M2). All models used species-specific landings from 2009 to 2018. Three models (M1, M2, M4) used biomass indices based on the exploitable biomass (individuals ≥ 50 cm). Moreover, total biomass indices were used (M3) to test if differences in landing observation error (LOE) terms were caused by a size-specific retention pattern. Finally a short time series model (M4), based on data from 2009 to 2018 only, was run to estimate multispecific catch data contribution to model outputs. All models were run using 1000000 iterations, with a burn-in of 500000 iterations and a thin of 1000. The best model was selected using the DIC (Divergence Information Criterion).

Table 3 Four models' scenarios (M1, M2, M3, M4) priors, *Y0* initial biomass in the first year of model run.

M1: long time series (Catches), vulnerable biomass indices (Survey) and depleted stocks (Y0)

M2: long time series (Catches), vulnerable biomass indices (Survey) and less depleted stocks (Y0)

M3: long time series (Catches), total biomass indices (Survey) and depleted stocks (Y0)

M4: short time series (Catches), vulnerable biomass indices (Survey) and depleted stocks (Y0)

Run	Species	Catches	Survey	Y0~Beta mode, SD
M1	*Raja clavata*	1990-2018	1990-2018 vulnerable biomass	0.3, 0.2
	Raja brachyura			0.3, 0.2
	Raja montagui			0.3, 0.2
M2	*Raja clava*	1990-2018	1990-2018 vulnerable biomass	0.5, 0.2
	Raja brachyura			0.5, 0.2
	Raja montagui			0.5, 0.2
M4	*Raja clavata*	1990-2018	1990-2018 total biomass	0.5, 0.2
	Raja brachyura			0.5, 0.2
	Raja montagui			0.5, 0.2
M5	*Raja clavata*	2009-2018	2009-2018 vulnerable biomass	0.5, 0.2
	Raja brachyura			0.5, 0.2
	Raja montagui			0.5, 0.2

2.3. Statistic test rebuilding speed

A Spearman correlation test was used to compare the relative biomass trends and rebuilding speed across species. A generalized least square regression was then used, with weights based on the standard deviation, to model relative biomass variations relative to years and species. The coefficients were extracted to compare the trends between species. Finally, significant contrast between species trends was tested using *lstrends* from *lsmeans* package (R version 3.6.3).

2.4. Selling prices

Species specific selling prices in euros.kg^{-1}were obtained from governmental (Belgium and Netherlands) fishing organisations (France) for three of the main countries involved in the ECNS Rajidae fisheries. However, the time series length, ten years, due to the late species specific identification, did not allow statistical test on prices trends. Furthermore the heterogeneity of the sources and data did not allow comparison between countries.

3. Result

3.1. Survey indices

Table 4 shows the differences in catchability between the BTS, and NS-IBTS and CGFS surveys for skates in the ECNS. For all species and biomass combinations, except for the vulnerable biomass index of the CGFS for *R. brachyura*, the estimated coefficients are significantly (p value <0.05) smaller than 0, implying that the BTS survey has a higher catchability compared to the NS-IBTS and CGFS surveys. For each combination of survey and skate species, the difference in catchability of the NS-IBTS and CGFS surveys with respect to the BTS survey is always larger for the total biomass than for the vulnerable biomass (Table 4). This may indicate that the BTS survey catches relatively more individuals < 50 cm than the other surveys.

Table 4 Summary statistics of the catch efficiency estimation of the NS-IBTS and CGFS with respect to the BTS survey for skate species in the ECNS.

			n	Est. coefficient (s.e.)	*p value*	raising factor
R. brachyura	NS-IBTS	≥ 50 cm	75	-0.178 (0.059)	0.003	1.20
		≥ 0 cm	145	-0.665 (0.051)	<0.001	1.94
	CGFS	≥ 50 cm	79	0.009 (0.060)	0.883	0.99
		≥ 0 cm	125	-0.438 (0.053)	<0.001	1.55
R. clavata	NS-IBTS	≥ 50 cm	412	-1.582 (0.016)	<0.001	1.82
		≥ 0 cm	533	-1.728 (0.013)	<0.001	2.64
	CGFS	≥ 50 cm	297	-0.598 (0.015)	<0.001	4.87
		≥ 0 cm	332	-0.970 (0.013)	<0.001	5.63
R. montagui	NS-IBTS	≥ 50 cm	335	-1.698 (0.030)	<0.001	8.37
		≥ 0 cm	533	-1.901 (0.026)	<0.001	9.22
	CGFS	≥ 50 cm	69	-2.124 (0.105)	<0.001	5.46
		≥ 0 cm	164	-2.221 (0.089)	<0.001	6.69

The leave-one-out analysis of the NS-IBTS Q1 and Q3 surveys indicate no problems for *R. clavata*. Both leave-one-out indices are very similar, and stay within the 95% confidence interval of the index based on all surveys for the entire time series. For *R. brachyura* and *R. montagui*, the Q3 and Q1 leave-one-out indices have a similar overall trend, although the index excluding the NS-IBTS-Q1 data is much more erratic during the last 10 years of the analysis. For both *R. brachyura* and *R. montagui*, exclusion of the NS-IBTS-Q1 data results in an upwards shift of the index during the final years (Figure 1). This is likely caused by the low coverage of the surveys, in particular the NS-IBTS-

Q3, in the North Western part of the North Sea (ICES rectangles 47E7, 47E8, 48E7, 48E8), an area where the *R. brachyura* and *R. montagui* are abundant (Daan *et* al., 2004; Heessen *et al*., 2015, S2), and the absence of other surveys in this area (i.e. BTS). The patchy distribution of the species may cause that in some years catches are high, in case the survey hits a hotspot, while in other years, the catches are low (ICES, 2020). Obviously, in case of a low sampling frequency, this will likely result in a more erratic trend.

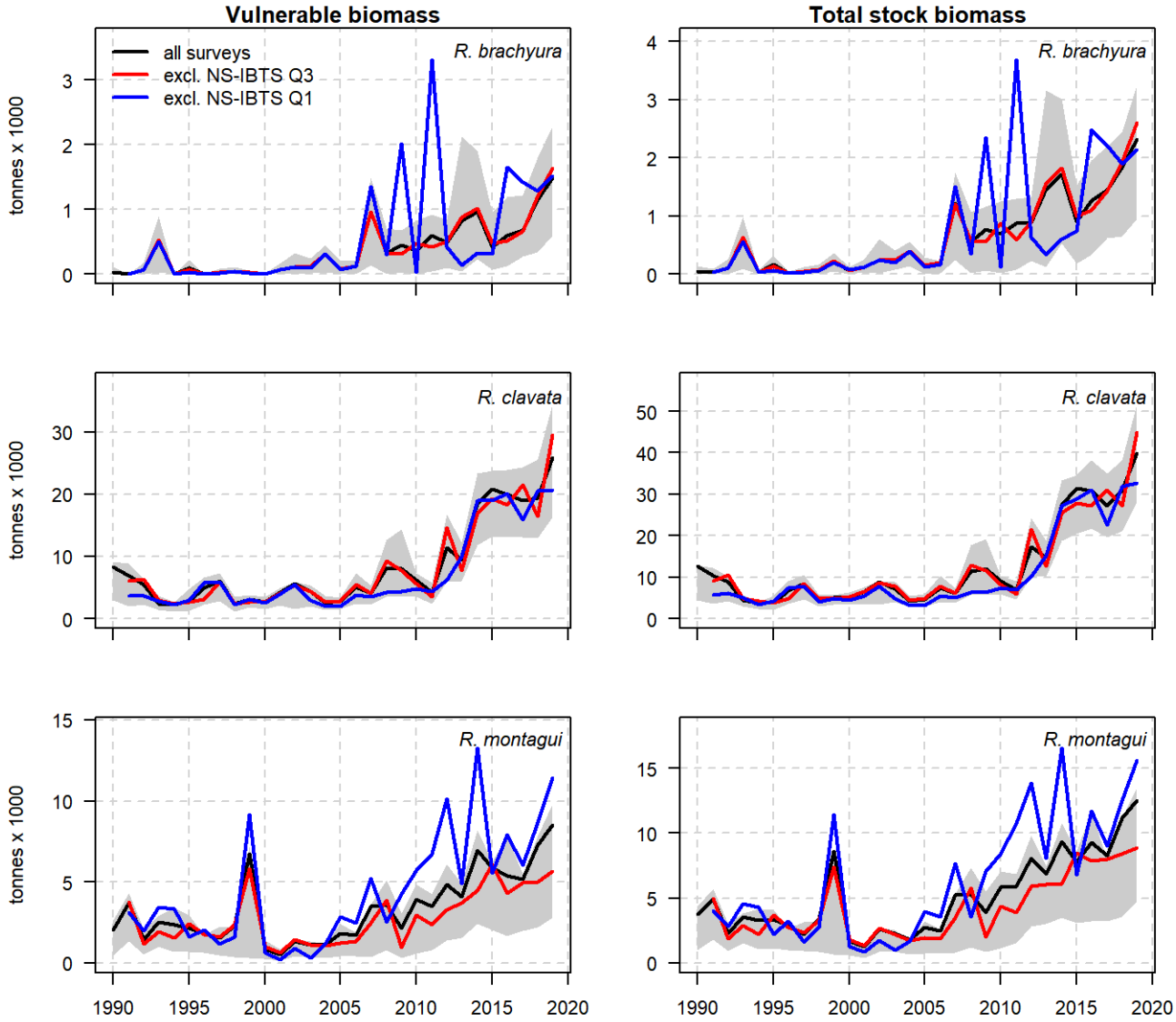


Figure 1 Survey indices and NS-IBTS Q1/Q3 leave-one-out indices for *R. brachyura* (upper panels), *R. clavata* (middle panels), and *R. montagui* (lower panels) by length bin (left panels: Vulnerable biomass >= 50 cm, right panels: The grey shade indicates the 95% confidence interval of the combined index using all survey data.

3.2. Catches

Rajidae landings from 1990 to 2008 decreased from 4444 to 2708 tonnes (Figure 2). Since 2009 the reported landings have started a slow increase from 1688 in 2009 to 2827 tonnes in 2018. Landings, when considering the three main species only, are predominantly composed of *R. clavata* (67- 82% of the total landings), *R. montagui* is the second most abundant in the total landings (9-23%), finally *R. brachyura* is the less abundant in landings (8-11%).

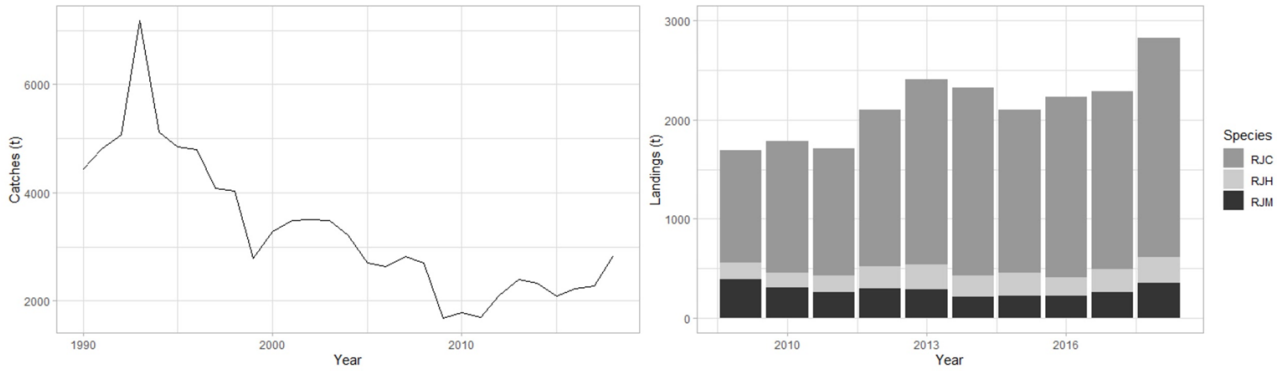


Figure 2 Landings (in tonnes) from ICES Historical Nominal Catches (1990-2008) and WGEF landings table (2009-2018) all species considered and WGEF landings table (2009-2018) RJC (*R.clavata*), RJH (*R. brachyura*), RJM (*R. montagui*)

To account for discard data heterogeneity, data correction was applied. Multiple regressions resulted in the presence of significant relations (p value < 0.001) for multi-rig otter trawl (OTT-DEF) for the three species. Bottom otter trawl (OTB-DEF) effort was significant for *R. clavata* only (p value < 0.001). The most discarded species were *R. clavata* and *R. montagui*, with a mean discard amounting to 1013 tonnes and 683.8 tonnes along the 2009-2018 time period. The less discarded species, *R. brachyura*, presented a mean discard amount of 226 tonnes (Figure 3). Considering discards in terms of additional mortality (0.5 of the discarded amount), they represent a mean value of 24 % of the total fishing mortality for *R. clavata*, 31 % for *R. brachyura* and 53 % for *R. montagui*.

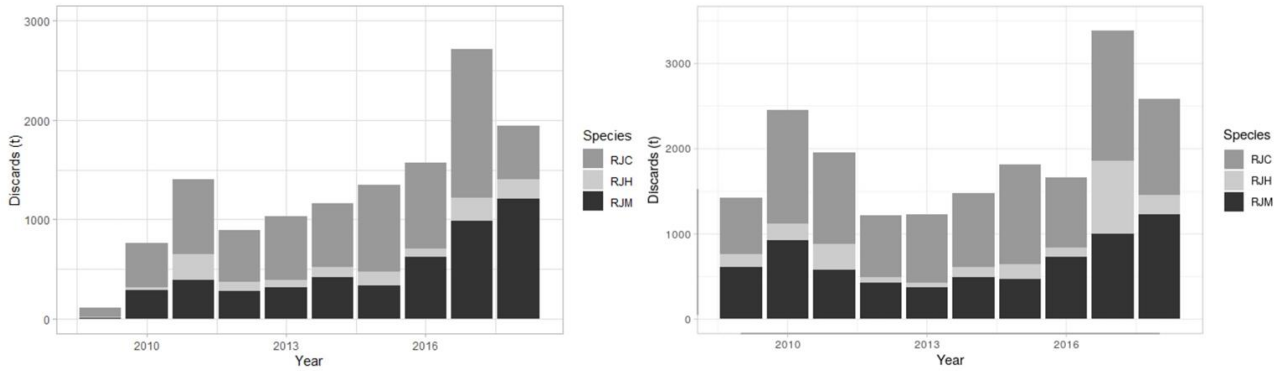


Figure 3 From left to right, raw discards (in tonnes) from WGEF discards table (2009-2018); discards estimations (in tonnes) from multiple regressions, discards were elevated by effort per métier (2009-2018); RJC (*R.clavata*), RJH (*R. brachyura*), RJM (*R. montagui*)

3.3. Model outputs

All models convergence were tested using Gelman and Rubin's test. The best performing model was M2 (Table 5).

Table 5 Number of parameters non fitting Gelman and Rubin's convergence diagnostic outputs at 1.05 and 1.1 threshold values per models, pD effective number of parameters, DIC Deviance Information Criterion

Model	Gelman Rubin's 1.05	Gelman Rubin's 1.1	pD	DIC
M1	71	0	941.6	2168.226
M2	0	0	921.3	2147.806
M3	69	0	931.6	2150.104
M4	0	0	880.7	2283.704

Prior posterior distributions for the carrying capacity, intrinsic growth rate and initial biomass are almost similar for all models (Figure 4). However, the use of semi informative priors for the *R. clavata* carrying capacity, to help model convergence, lead to a truncated posterior distribution.Despite the use of different sets of priors and data, all models gave similar parameters estimations (Table 6), suggesting that the models are mainly driven by the data. Furthermore, the short scenario (M4) gave similar outputs, highlighting the importance of the ten years of species-specific data contribution. Moreover, in all cases, models showed an initial relative biomass of around 0.225 (0.198-0.228) for *R. clavata*, 0.052 (0.041-0.071) for *R. brachyura* and 0.088 (0.066-0.115) for *R. montagui*. Clear differences in terms of landing observation errors appeared for *R. montagui*, being 2 times superior to the landing observation errors of the two other species. However, discard observation errors were around 0.004 for all species.

Figure 4 Prior posterior distribution for the three chains of the four models; K (carrying capacity); r (intrinsic growth rate); Y (Initial biomass in at the beginning of the model), depleted scenarios (black lines), non depleted scenario (grey lines); 1 (*R. clavata*), 2 (*R. brachyura*), 3 (*R. montagui*)

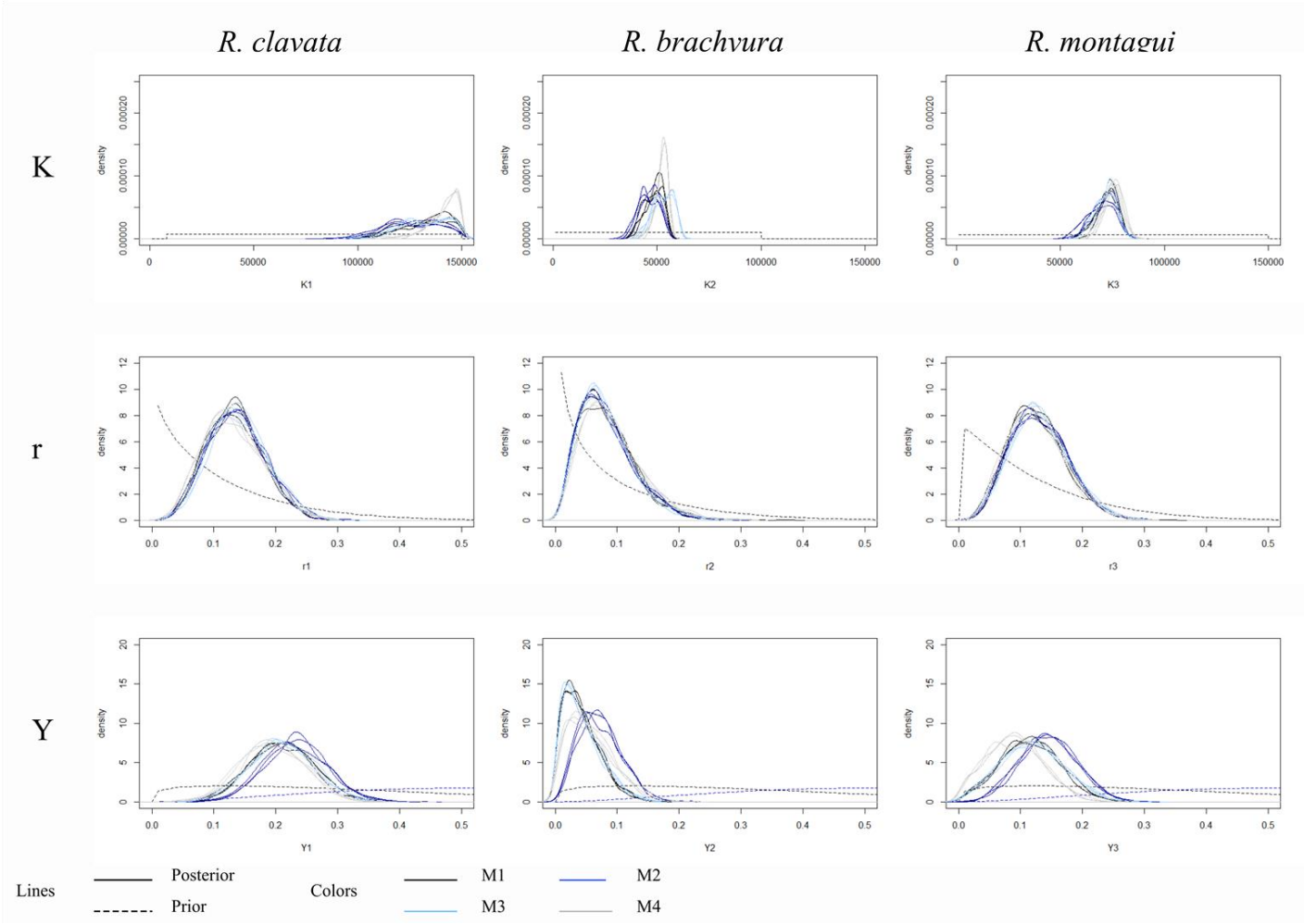


Table 6 For each models scenarios (M1, M2, M3, M4) mean parameter estimations per species and models, *K* carrying capacity, *r* intrinsic growth rate, *Y0* initial biomass in the first year of model run, *LOE* observation errors in species-specific landings proportions, *DOE* observation errors in species-specific discards proportions, *q* catchability, *σ* process error and *τ* observation error for biomass indices.

	M1	**M2**	**M3**	**M4**
R. clavata				
K	132 850	127 037	132 472	141 618
r	0.149	0.151	0.157	0.137
Y0	0.229	0.246	0.228	0.198
LOE	0.003	0.003	0.003	0.003
DOE	0.003	0.003	0.003	0.003
R. brachyura				
K	48 402	46 261	53 098	52 275
r	0.081	0.082	0.083	0.088
Y0	0.042	0.071	0.041	0.053
LOE	0.003	0.003	0.003	0.003
DOE	0.004	0.004	0.004	0.004
R. montagui				
K	110 118	106 189	110 504	120 812
r	0.106	0.106	0.106	0.115
Y0	0.085	0.115	0.086	0.066
LOE	0.009	0.009	0.009	0.009
DOE	0.005	0.005	0.005	0.005
All species				
q	0.0003	0.0004	0.0003	0.0004
σ	0.050	0.050	0.050	0.050
τ	0.003	0.003	0.003	0.003

3.4. Biomass trajectories

Biomass trajectories were analysed based on the best performing model, M2. All species stocks are currently rebuilding between 1990 and 2018, from 0.47 of B_{MSY} to 1.26 for *R. clavata*, 0.01 to 0.20 for *R. brachyura* and 0.16 to 0.45 for *R. montagui* (Figure 5). All stocks were considered as depleted in 1990, *R. clavata* is the only stock estimated to reach the biomass at MSY at the end of the time series. The Spearman correlation tests indicated a clear correlation between time and biomass for all species ($p < 0.001$) (Table 7). *R. clavata* presented the fastest trend using a generalized least square regression (0.042-0.073), followed by *R. montagui* (0.014-0.030), and *R. brachyura* (0.008-0.013). All long time series models (1990-2018) trends contrasts were significant between *R.clavata* and *R. brachyura* (p value <0.001), as well as between *R. clavata* and *R. montagui* (p value < 0.001). No

significant contrasts in trends were highlighted between *R. brachyura* and *R. montagui* (p value 0.667-0.740).

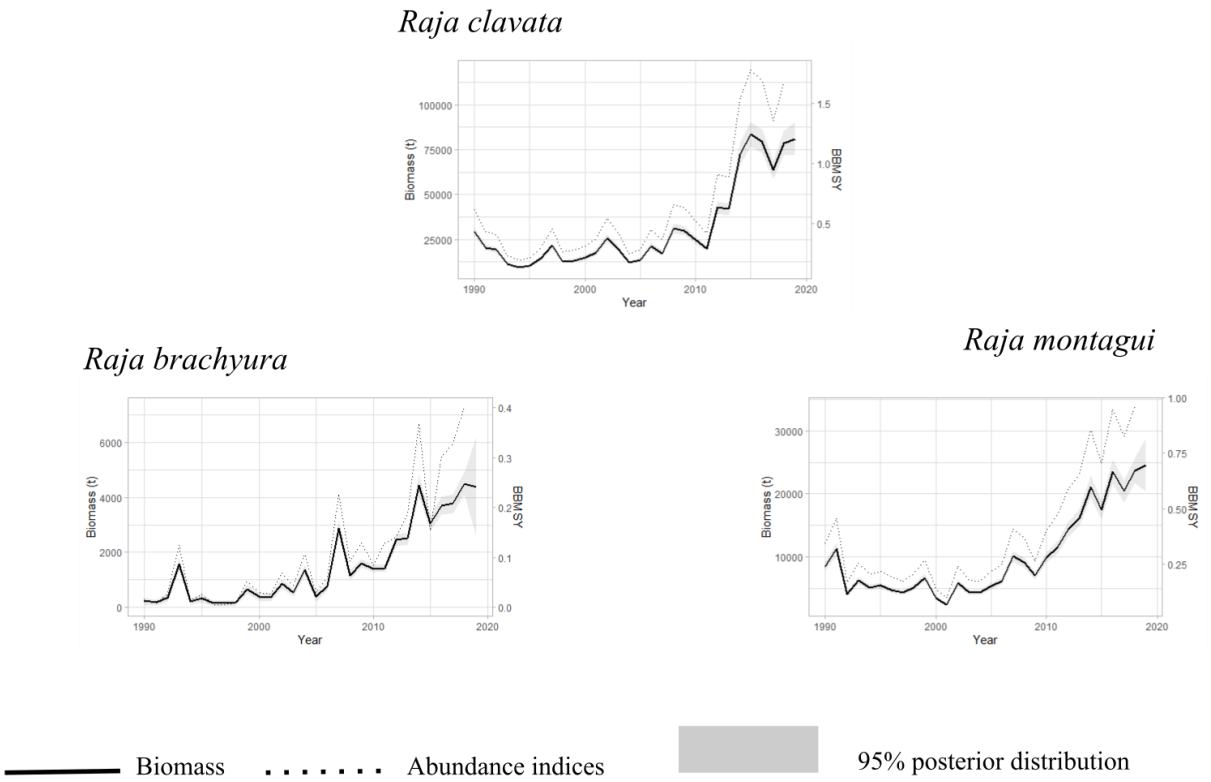


Figure 5 Absolute (Biomass (t)) and relative biomass (BMSY) estimations from M2 (1990-2019) prediction. The black line indicates mean estimate. The grey area around the black line indicates the 95% distribution. The dotted grey line indicates the abundance index input, on the scale of the absolute biomass estimates.

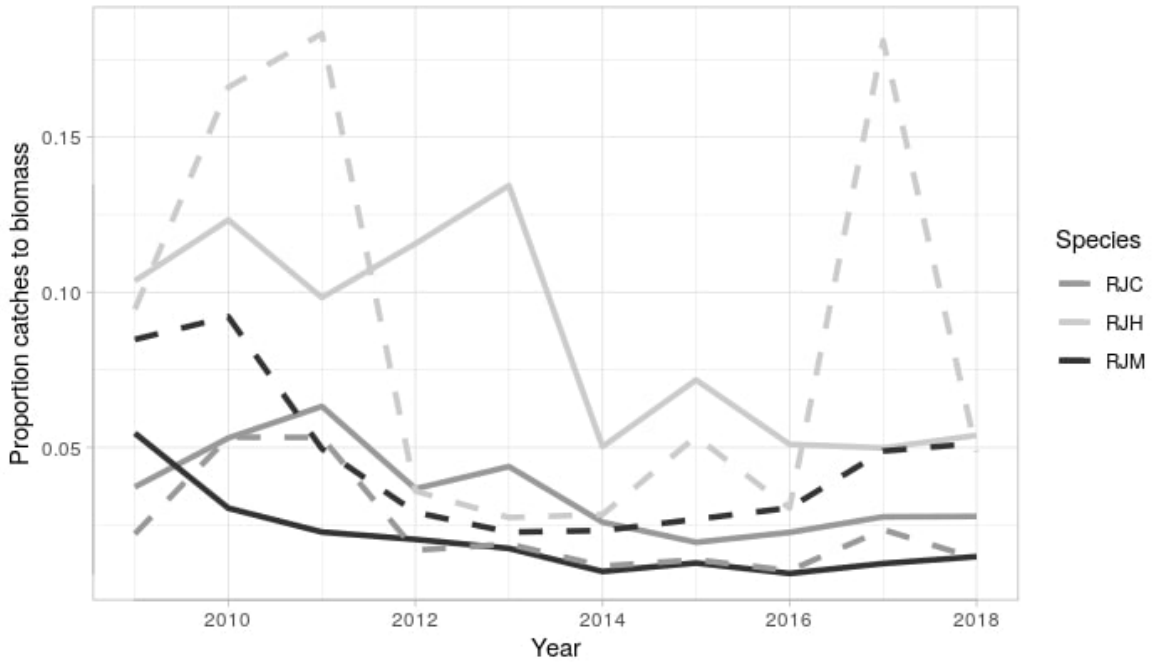
Table 7 For each models scenarios (M1, M2, M3, M4)**:** Spearman correlation test *p value* between biomass and year; Generalized least square weight regression trend results; Trend contrasts estimates and *p values* per species pairs, RJC (*R. clavata*), RJH (*R. brachyura*), RJM (*R. montagui*).

		M1	**M2**	**M3**	**M4**
Biomass time series length		1990-2018	1990-2018	1990-2018	2009-2018
Spearman correlation					
RJC	p value	<0.001	<0.001	<0.001	0.004
RJH		<0.001	<0.001	<0.001	<0.001
RJM		<0.001	<0.001	<0.001	<0.001
Generalised least square					
RJC	trends	0.042	0.043	0.044	0.073
RJH		0.008	0.008	0.008	0.013
RJM		0.014	0.014	0.015	0.030
Trend contrasts					
RJC-RJH	contrast estimate	0.034	0.035	0.037	0.060
	p value	<0.001	<0.001	<0.001	0.020
RJC-RJM	contrast estimate	0.028	0.029	0.030	0.043
	p value	<0.001	<0.001	<0.001	0.028
RJH-RJM	contrast estimate	-0.006	-0.006	-0.007	-0.016
	p value	0.707	0.699	0.667	0.740

3.5. Discards as a result of a sum of factors

During the 2009 to 2018 period the relative amounts of discards and landings to biomass were higher for *R. brachyura* (3-18% and 5-13%) than *R. clavata* (1-5% and 2-6%) and *R. montagui* (2-9% and 1-5%) (Figure 6). The amount of discards to biomass is higher than the amount of landings to biomass during the entire period for *R. montagui* (1 to 6% of difference). Contrariwise the amount of discards to biomass is lower than the amount of landings to biomass for *R. clavata* (0 to 2% of difference). This difference presents a more eclectic pattern for *R. brachyura ,* the relative discard amount was higher until 2011 and lower since then, apart from 2017.

Figure 6 From the left to the right. Proportion of discards (dotted lines) and landings (solid lines) to biomass per species. RJC (*R. clavata*), RJH (*R. brachyura*), RJM (*R. montagui*).



Few information was available on species specific prices. According to Fishing Organisations reports from Belgium and Netherlands the most expensive species within the complex was *R. brachyura* (Table 8). According to french data the variability between species prices were less substantial, and the most valuable species differ depending on the year.

Table 8 Species prices in euros.kg^{-1} per country in 2009, 2014 and 2018, FRA (France), BEL (Belgium), NDL (Nederlands)

Countries	Species	Prices (euros.kg^{-1})			Sources
		2009	2014	2018	
BEL	*R. clavata*	1.96	2.63	2.17	Quovis Database (Department of Agriculture and Fisheries, accessed on 22 September 2020)
	R. brachyura	2.59	3.18	2.66	
	R. montagui	1.92	2.19	1.97	
FRA	*R. clavata*		2.39	2.26	France AgriMer, 2019
	R. brachyura	2.69	2.8	2.83	
	R. montagui	2.93	3.07	2.25	
NDL	*R. clavata*	2.01	1.74	1.39	Netherlands Enterprise Agency (RVO, accessed September 2020)
	R. brachyura		2.81	2.63	
	R. montagui	2.69	2.63	2.22	

4. Discussion

4.1 Landings, discards and biomass dynamics

Data analysis coupled with model outputs, suggests an overfishing event of Rajidae stocks preceding 1990 first management measures. Stocks stayed depleted until the implementation of 2009 management measures that encompass the whole ECNS area. Rajidae stock declines during the 20th century period in the North East Atlantic has already been reported in surrounding fishing areas (Dulvy et al., 2000, Manrandel et al., 2019). Stock biomass trends indicated, for all species of the complex, a common rebuilding dynamic. A similar increase in recent years has been observed for the same species in the Bay of Biscay (Marandel et al., 2019).

Species within the Rajidae complex present disparities in their rebuilding dynamics and speed. The initial biomass in 1990 appeared to be one of the main drivers for those speed differences, with the less depleted stock for *R. clavata* rebuilding faster and presenting a significant contrast with *R. brachyura* and *R. montagui*. Moreover, several large scale studies have already been conducted to evaluate global sensitivity drivers in Rajidae. Life History Parameters (LHP) including growth and maturity, differ within the Rajidae complex species, *R. brachyura* being the largest species and maturing the latest (Dulvy et al., 2000). Body size is usually identified as one of the main factors (Dulvy & Reynolds, 2002) that could impact Rajidae sensitivity to extinction risk. This pattern of sensitivity have already been highlighted in north-western North Sea classifying, from the most to the least sensitive species, *R. clavata* (L_∞ 85cm), *R. montagui* (L_∞ 75cm), *L. naevus* (L_∞ 70cm) and *R. radiata* (L_∞ 60cm) (Walker & Hislop, 1996). Though, body size might have an important impact on sensitivity, other factors might affect the extinction risk at the genus level such as the reproductive mode (Garcia et al., 2008). Further analysis of the different parameters outputs put into light disparities in landings and discards patterns. LOE (Landing Observation Error) heterogeneity between species demonstrated variabilities in observed landings and estimated biomass trends relations. The highest LOE is obtained for *R. montagui*, the less landed species and less valuable one. This important LOE might not be caused by misidentification between *R. brachyura* and *R. montagui* only since *R. brachyura* LOE is low and might concern misreport issues specific to *R. montagui*. Retention patterns appear clearly within the Rajidae complex, *R. brachyura* being preferred, with the most important landing to total catch ratio, to *R. clavata* and *R. clavata* to *R. montagui*. No causality could be identified to explain the relation between retention patterns and biomass dynamics. The lack of species specific data, before the implementation of management procedures, complicated the identification of the retention phenomena. On one hand, the retention patterns could be the cause of a severely depleted stock; and on the other hand, they might have impacted the rebuilding speed solely. In both situations, the correlation between stock status and retention patterns have to be analyzed further to identify the most vulnerable species within a complex, according to their past fishing history.

4.2. Retention pattern causes

The ECNS Rajidae complex case of study do not provide length data, thus, in the following paragraph retention pattern refer to the relation between the proportion of Rajidae caught that are kept on board to be landed and the proportion discarded at sea. The use of species specific assessment model, including a landing and a discard compartments allow the estimation of the relation between biomass, landings and discards quantities estimated by the model. These metrics helped the identification of species exposed to the highest fishing mortality within the complex. The evidence of differences in retention patterns between species will foster further research to understand variables driving these patterns. Furthermore, it highlighted current management measures weaknesses. Retention pattern drivers within a complex are not always clearly understood (Feekings et al., 2012). Many factors, such as the relative economic value or cultural habits, could participate in these preferences (Catchpole et al., 2011). However, it is impossible to acquire a universal picture of these preferences since the factors are affected by regional and cultural variations through time (Olsen et al., 2008) as

well as resource availability (Damalas et al., 2015). No uniformed minimum landing size is currently implemented in the whole study area. Yet, country specific minimum landing sizes are present for Belgium (50cm total length), France (45cm total length), United Kingdom (40cm total width) and Netherlands (55cm total length). A unique minimum landing size for all species favoured the retention of bigger species (Silva et al., 2012). The exploitable biomass index (individuals over 50cm of total length only) integrate this specificity, by defining an exploitable biomass corresponding to the mean of minimum landings size expressed in total length. However, if differences in retention patterns were caused by size differences only, one might expect that using under commercial size indices would increase the LOE for all species, from the smallest to the largest, yet no clear differences between M3, full abundance indices inputs, and the other models', commercial size abundance indices, outputs confirmed this hypothesis. Furthermore, body size differences were expected to impact species specific price, since the size and thickness of the wings are proportional to their total length. Within the ECNS Rajidae complex, *R. brachyura* is the largest species. Since only the wings are sold for Rajidae, the size heterogeneity between species might result in an economical preference for *R. brachyura*.

Price differences were observed during this study, *R. brachyura* being the most expensive species in Belgium and Netherlands. Differences in prices might be caused by other factors than size such are public preferential demand or habits (Claret et al., 2012). The co-occurrence between the implementation of a more constraining in TAC, almost reached every year since 2013, and the change in landings to biomass and discards to biomass ratio in 2012 for *R. brachyura* highlighted the potential impact of management measures on retention patterns. Changes in retention pattern have been observed for other species, such as *Lamna nasus*, *Squalus acanthias* and *Mustelus asterias* in UK fisheries simultaneously with an increase in management measures (Silva & Ellis, 2019). In this example the constraining TAC of species managed individually, *L. nasus* and *S. acanthias*, resulted in an increase of their discards, when the non quota species, *M. asterias*, landings increased. Management measures at the complex level could result in slightly different strategies since fishermen chose which species they are willing to land. Economical motives might led to an an increase of the less commercially important species discards to favour the higher economical value species. In the context of the ECNS Rajidae complex the rebuilding of the stocks associated to a constraining TAC might enhance the landings of the most valuable species, *R. brachyura*, and an increase in the other species discards.

All of the different factors affecting the retention pattern have to be understood to assess species specific vulnerability. Large scale studies usually do not take into consideration the relative vulnerability of species within a complex or a multispecific fishery (Riginella *et al.*, 2020). In these situations, Rajidae' vulnerability is not only dependent on their own species-specific life history traits, but also of their relative vulnerability to the species they are managed with. Consequently, the largest species within a complex suffer from a compounded vulnerability linked to their life history sensitivity and their preferential retention.

4.3. Chondrychtians assessment and management

The identification of retention patterns, in complex management, questions potential methodology applied to assess Rajidae stocks. In the North East Atlantic more than one chondrichthyan stock is managed or even assessed as a complex. Complex assessments are usually linked to identification difficulties that cause inaccurate species specific reports such as smoothhound, *Mustelus Spp.* (Farrell et al., 2009), common skate complex, *Dipturus batis* and *Dipturus intermedius* (Iglésias et al., 2010) in the north east Atlantic. For all of them, the only catch data currently used in TAC recommendations are landings (ICES, 2019a). The growing requirements for stock assessments imply a fast evolution in most assessment methodologies. When choosing new tools for sharks and rays assessments, discard and landing relations have to be taken into consideration (James et al., 2016, Pennino *et al.*, 2017). Indeed, Catch Only Methods (COM) are frequently applied to DLS (Anderson et al., 2017). When COMs are used, strong assumptions are made on the implementation of discards. In the Rajidae

complex case discard report substantially increased the 2009-2018 period. This disparity was presumably linked to an underestimation of the discards during most of the time series, caused by heterogeneity in observer programs data for Rajidae species. It is clear that some species' total catch are mainly composed of discards when others relied on landings only, even if the commercial interest for the species seems equivalent. Multispecies production model including discards and landings compartments give insights on these disparities' identification (Hollowed et al., 2000). In the case of the Rajidae complex, multispecies model also provide the opportunity to use longer time series of catch, by dealing with the heterogeneity in identification. Other surplus production models could be tested on the species specific time series (10 year), such as SPicT (Pedersen & Berg, 2017) or CMSY (Martell & Froese, 2013), but their precision as well as their sensitivity to data inputs will be higher. When catch data issues are too important for chondrichthyans, other models based on life history parameters have been developed (Bradshaw et al., 2018, Free et al., 2020, Walsh et al., 2018). Comparing different models outputs should in any case help the identification of the best method available to follow these stocks trends(Carruthers et al., 2014). However, the combination of different type of models through ensemble approach would enable to go a step further and use all pertinent model together (Anderson et al., 2017, Walsh et al., 2018).

Conclusion

ECNS Rajidae fishery presents clear retention patterns, but further studies are needed to understand which biological, economical and sociological processes are behind these patterns. However, within the complex, retention preferences appeared to affect the stocks' rebuilding dynamics.
Retention patterns differences in bycatch DLS implies constraints in stock assessment models. These constraints impact catch data reliability and model choices. In the case of complex management, a multispecies model approach might give insights on the appropriate models. In other cases, when retention patterns are not identified and discards data sparse, catch only models should be avoided and parallel life history approach considered. Most exploited chondrichthyans stocks fall within those constraints and would encounter the same management issues.

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References

Anderson, S. C., Cooper, A. B., Jensen, O. P., Minto, C., Thorson, J. T., Walsh, J. C., ... & Osio, G. C. (2017). Improving estimates of population status and trend with superensemble models. *Fish and Fisheries*, *18*(4), 732-741.

Batsleer, J., Hamon, K.G., van Overzee, H.M.J., Rijnsdorp, A.D., Poos, J.J. (2015) High-grading and over-quota discarding in mixed fisheries. *Reviews in Fish Biology and Fisheries*, 25(4), 715-736.

Benoît, H. P., Plante, S., Kroiz, M., & Hurlbut, T. (2013). A comparative analysis of marine fish species susceptibilities to discard mortality: effects of environmental factors, individual traits, and phylogeny. *ICES Journal of Marine Science*, *70*(1), 99-113.

Bradshaw, C. J., Prowse, T. A., Drew, M., Gillanders, B. M., Donnellan, S. C., & Huveneers, C. (2018). Predicting sustainable shark harvests when stock assessments are lacking. *ICES Journal of Marine Science*, *75*(5), 1591-1601.

Borges, L., Cocas, L., & Nielsen, K. N. (2016). Discard ban and balanced harvest: a contradiction?. *ICES Journal of Marine Science*, *73*(6), 1632-1639.

Carruthers, T. R., Walters, C. J., & McAllister, M. K. (2012). Evaluating methods that classify fisheries stock status using only fisheries catch data. *Fisheries Research*, *119*, 66-79.

Carruthers, T. R., Punt, A. E., Walters, C. J., MacCall, A., McAllister, M. K., Dick, E. J., & Cope, J. (2014). Evaluating methods for setting catch limits in data-limited fisheries. *Fisheries Research*, *153*, 48-68.

Catchpole, T. L., Enever, R., Maxwell, D. L., Armstrong, M. J., Reese, A., & Revill, A. S. (2011). Constructing indices to detect temporal trends in discarding. *Fisheries Research*, *107*(1-3), 94-99.

Claret, A., Guerrero, L., Aguirre, E., Rincón, L., Hernández, M. D., Martínez, I., ... & Rodríguez-Rodríguez, C. (2012). Consumer preferences for sea fish using conjoint analysis: Exploratory study of the importance of country of origin, obtaining method, storage conditions and purchasing price. *Food Quality and Preference*, *26*(2), 259-266.

Clarke, M. W. (2009). Sharks, skates and rays in the northeast Atlantic: population status, advice and management. *Journal of Applied Ichthyology*, *25*, 3-8.

Cortés, E., Brooks, L., & Scott, G. (2002). Stock assessment of large coastal sharks in the US Atlantic and Gulf of Mexico. Panama City, Florida: National Marine Fisheries Service, Southeast Fisheries Science Center.

Costello, C., Ovando, D., Hilborn, R., Gaines, S. D., Deschenes, O., & Lester, S. E. (2012). Status and solutions for the world's unassessed fisheries. *Science*, *338*(6106), 517–520. https://doi.org/10.1126/ science.1223389

Daan, N., Heessen, H. J. L., & ter Hofstede, R. (2005). North Sea Elasmobranchs: distribution, abundance and biodiversity. ICES.

Damalas, D., Maravelias, C. D., Osio, G. C., Maynou, F., Sbrana, M., Sartor, P., & Casey, J. (2015). Historical discarding in Mediterranean fisheries: a fishers' perception. *ICES Journal of Marine Science*, *72*(9), 2600-2608.

Davidson, L. N., Krawchuk, M. A., & Dulvy, N. K. (2016). Why have global shark and ray landings declined: improved management or overfishing?. *Fish and Fisheries*, *17*(2), 438-458.

Davis, M. W. (2002). Key principles for understanding fish bycatch discard mortality. *Canadian Journal of Fisheries and Aquatic Sciences*, *59*(11), 1834-1843.

Dulvy, N. K., Metcalfe, J. D., Glanville, J., Pawson, M. G., & Reynolds, J. D. (2000). Fishery stability, local extinctions, and shifts in community structure in skates. *Conservation Biology*, *14*(1), 283-293.

Dulvy, N. K., & Reynolds, J. D. (2002). Predicting extinction vulnerability in skates. *Conservation Biology*, *16*(2), 440-450.

Dulvy, N. K., Fowler, S. L., Musick, J. A., Cavanagh, R. D., Kyne, P. M., Harrison, L. R., ... & Pollock, C. M. (2014). Extinction risk and conservation of the world's sharks and rays. *elife*, *3*, e00590.

Enever, R., Catchpole, T. L., Ellis, J. R., & Grant, A. (2009). The survival of skates (Rajidae) caught by demersal trawlers fishing in UK waters. *Fisheries Research*, *97*(1-2), 72-76.

FAO (2020) The State of World Fisheries and Aquaculture 2020. Sustainability in action. Rome. 224P

Farrell, E. D., Clarke, M. W., & Mariani, S. (2009). A simple genetic identification method for Northeast Atlantic smoothhound sharks (Mustelus spp.). *ICES Journal of Marine Science*, *66*(3), 561-565.

France AgriMer (2019). Données de vente déclarées en halles à marée en 2019. 96pp.

Feekings, J., Bartolino, V., Madsen, N., & Catchpole, T. (2012). Fishery discards: factors affecting their variability within a demersal trawl fishery. *PloS one*, *7*(4), e36409.

Free, C. M., Jensen, O. P., Anderson, S. C., Gutierrez, N. L., Kleisner, K. M., Longo, C., ... & Walsh, J. C. (2020). Blood from a stone: Performance of catch-only methods in estimating stock biomass status. *Fisheries Research*, *223*, 105452.

García, V. B., Lucifora, L. O., & Myers, R. A. (2008). The importance of habitat and life history to extinction risk in sharks, skates, rays and chimaeras. *Proceedings of the Royal Society B: Biological Sciences*, *275*(1630), 83-89.

Geng, Z., Punt, A. E., Wang, Y., Zhu, J., & Dai, X. (2020). On the dangers of including demographic analysis in Bayesian surplus production models: A case study for Indian Ocean blue shark. Fisheries Research, 230, 105636.

Heessen, H. J., Daan, N., & Ellis, J. R. (Eds.). (2015). *Fish atlas of the Celtic Sea, North Sea, and Baltic Sea: Based on international research-vessel surveys*. Wageningen Academic Publishers.

Hilborn, R., Amoroso, R. O., Anderson, C. M., Baum, J. K., Branch, T. A., Costello, C., ... & Kurota, H. (2020). Effective fisheries management instrumental in improving fish stock status. *Proceedings of the National Academy of Sciences*, *117*(4), 2218-2224.

Hollowed, A. B., Bax, N., Beamish, R., Collie, J., Fogarty, M., Livingston, P., ... & Rice, J. C. (2000). Are multispecies models an improvement on single-species models for measuring fishing impacts on marine ecosystems?. *ICES Journal of Marine Science*, *57*(3), 707-719.

ICES (2003) Report of the planning group on commercial catch, discards and biological sampling (PGCCDBS), 4-7 March 2003, Rome, Italy. ICES CM 2003/ACFM:16. 40pp.

ICES (2017) Report of the Working Group on Mixed Fisheries Advice (WGMIXFISH ADVICE), 22–26 May 2017, Copenhagen, Denmark. ICES CM 2017/ACOM:18. 128 pp.

ICES (2019a). Report of the Working Group on Elasmobranch Fishes (WGEF). ICES CM. 964pp.

ICES. (2019b). Manual for the International Bottom Trawl Surveys. Series of ICES Survey Protocols. SISP 10, Version 10. 86 pp.

ICES. (2019c). Manual for the Offshore Beam Trawl Surveys, Version 3.4, April 2019, Working Group on Beam Trawl Surveys. 54pp

Iglésias, S. P., Toulhoat, L., & Sellos, D. Y. (2010). Taxonomic confusion and market mislabelling of threatened skates: important consequences for their conservation status. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *20*(3), 319-333.

James, K. C., Lewison, R. L., Dillingham, P. W., Curtis, K. A., & Moore, J. E. (2016). Drivers of retention and discards of elasmobranch non-target catch. *Environmental Conservation*, *43*(1), 3-12.

Karagiannakos, A. (1996). Total Allowable Catch (TAC) and quota management system in the European Union. *Marine Policy*, *20*(3), 235-248.

Kraak, S. B., Daan, N., & Pastoors, M. A. (2009). Biased stock assessment when using multiple, hardly overlapping, tuning series if fishing trends vary spatially. *ICES Journal of Marine Science*, *66*(10), 2272-2277.

Letaconnoux, R. (1948). Effets de la guerre sur la constitution des stocks de poissons. *Rapport et Proces-Verbaux des Reunions*, *122*, 55-62.

Marandel, F., Lorance, P., & Trenkel, V. M. (2019). Determining long-term changes in a skate assemblage with aggregated landings and limited species data. *Fisheries Management and Ecology*, *26*(4), 365-373.

Marchal, P., Andersen, J. L., Aranda, M., Fitzpatrick, M., Goti, L., Guyader, O., ... & Macher, C. (2016). A comparative review of fisheries management experiences in the European Union and in other countries worldwide: Iceland, Australia, and New Zealand. *Fish and Fisheries*, *17*(3), 803-824.

Martell, S., & Froese, R. (2013). A simple method for estimating MSY from catch and resilience. *Fish and Fisheries*, *14*(4), 504-514.

Musick, J. A., & Bonfil, R. (Eds.). (2005). Management techniques for elasmobranch fisheries (No. 474). Food & Agriculture Org..

Newell, R. G., Sanchirico, J. N., & Kerr, S. (2005). Fishing quota markets. *Journal of environmental economics and management*, *49*(3), 437-462.

Olsen, S. O., Heide, M., Dopico, D. C., & Toften, K. (2008). Explaining intention to consume a new fish product: A cross-generational and cross-cultural comparison. *Food quality and preference*, *19*(7), 618-627.

Van Overzee, H. M. J., Kraan, M. L., & Quirijns, F. J. (2014). *Beleidsondersteunend onderzoek haaien en roggen* (No. C009/14). IMARES.

Pedersen, M. W., & Berg, C. W. (2017). A stochastic surplus production model in continuous time. *Fish and Fisheries*, *18*(2), 226-243.

Pennino, M. G., Vilela, R., Valeiras, J., & Bellido, J. M. (2017). Discard management: A spatial multi-criteria approach. *Marine Policy*, *77*, 144-151.

Riginella, E., Correale, V., Marino, I. A., Rasotto, M. B., Vrbatovic, A., Zane, L., & Mazzoldi, C. (2020). Contrasting life-history traits of two sympatric smooth-hound species: implication for vulnerability. *Journal of Fish Biology*, *96*(3), 853-857.

Rochet, M. J., & Trenkel, V. M. (2005). Factors for the variability of discards: assumptions and field evidence. *Canadian Journal of Fisheries and Aquatic Sciences*, *62*(1), 224-235.

Schram, E., & Molenaar, P. (2018). *Discards survival probabilities of flatfish and rays in North Sea pulse-trawl fisheries* (No. C037/18). Wageningen Marine Research.

Silva, J. F., Ellis, J. R., & Catchpole, T. L. (2012). Species composition of skates (Rajidae) in commercial fisheries around the British Isles and their discarding patterns. *Journal of Fish Biology*, *80*(5), 1678-1703.

Silva, J. F., & Ellis, J. R. (2019). Bycatch and discarding patterns of dogfish and sharks taken in English and Welsh commercial fisheries. *Journal of fish biology*, *94*(6), 966-980.

Stevens, J. D., Bonfil, R., Dulvy, N. K., & Walker, P. A. (2000). The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. *ICES Journal of Marine Science*, *57*(3), 476-494.

Walker, P. A., & Hislop, J. R. G. (1996). Sensitive skates or resilient rays. *Spatial and temporal shifts in ray species composition in the Central and NW North Sea. ICES CM*, 1-11.

Walsh, J. C., Minto, C., Jardim, E., Anderson, S. C., Jensen, O. P., Afflerbach, J., ... & Selig, E. R. (2018). Trade-offs for data-limited fisheries when using harvest strategies based on catch-only models. *Fish and Fisheries*, *19*(6), 1130-1146.

Wetzel, C. R., & Punt, A. E. (2015). Evaluating the performance of data-moderate and catch-only assessment methods for US west coast groundfish. *Fisheries Research*, *171*, 170-187.

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., ... & Jensen, O. P. (2009). Rebuilding global fisheries. *science*, *325*(5940), 578-585.