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Broadening the scope to non-target and data-limited stocks.

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Zuzendariak / Directors

Dr. Dorleta García & Dr. Leire Ibaibarriaga

eman ta zabal zazu

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- Dr. **Claire Macher**, researcher at French Research Institute for Exploitation of the Sea, Ifremer (Nantes, France).
- Dr. **Marc Taylor**, researcher at the Thuenen Institute of Sea Fisheries (Braunschweig, Deutschland).

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Guraso,
anai-arreba,
eta Asier-eri

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Tesia ezerezetik gauzatzeko prozesu hau hain izan da luzea eta berria niretzat, badakidala ez nintzatekeela bukatzeko gai izango aldamenean izan dudan babes eta laguntzarik gabe. Honela, ez dut tesia bukatutzat eman nahi beraiei guztiei eskerrak eman gabe.

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Probatu ez duenak ez daki zer den hau!
Probatu ez duenak ez daki zer den hau!
Negu gorriak

Joxean Altuna Etxarri, egunero —
Querer es poder.

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Laburpena

Muturrekoa izan daitekeen seigarren biodibertsitate-krisian murgilduta gaude, eta ondorioz, gizakion osasunaren etorkizuna arriskuan dago. Gehiegizko arrantza-jarduera, itsasoko biodibertsitate galeraren zio nagusietako bat da. Kudeaketa estrategia jasangarriak, ondorioz, funtsezkoak dira itsasoko ekosistemen biodibertsitatea berroneratzeko edo mantentzeko. 1970az geroztik, Europar Batasuneko arrantza-baliabideen kudeaketa, arrantza politika erkidearen (Common Fisheries Policy, CFP) bitartez gauzatzen da. CFPren 2013ko azken berrikuspenean, kudeaketa helburua errendimendu jasangarri maximoan (Maximum Sustainable Yield, MSY) ezarri zen: honek, ahalik eta arrain-bolumenik handiena harrapatzea baimentzen du, beti ere populazioak ekoizpen maila maximoan mantendu behar direlarik. Honez gain, espezieen kudeaketak, zuhurtasunaren printzipioa ere bete behar du. Honen arabera, erabateko ziurtasun zientifiko ezak, ez luke ingurumenaren narriatzea geldi dezaketen kudeaketa neurrien ezarpena atzeratzeko arrazoia izan behar. Arrantza-flotak erreminta ezberdinen bitartez kudeatzen dira: baimenduta dagoen guztizko harrapaketa (Total Allowable Catch, TAC) neurritik hasita, lehorreratzen diren banakoen tamaina minimoak, sare-begien tamaina mugak eta babestutako itsas eremuak bezalako neurri teknikoetaraino. Harrapatutako arrainak portura eramateko obligazioa, CFPren azken erreformak ezarri zuen erregulazio garrantzitsuenetariko bat da. Erregulazio honen xedea da bazterkinak ezabatzea, ustekabeko edo nahi gabeko harrapaketak saihestuz eta murriztuz. Erregulazio honen arabera, harrapaketa-muga duten eta/edo legearen araberako erreferentziazko kontserbazio-tamaina minimo baten menpe dauden espezieen banako guztiak lehorreratuak izan behar dira harrapatuak izanez gero. Horrez gain, CFPren azken erreforman, urte anitzeko kudeaketa planak erreminta garrantzitsuenetariko bat bilakatu ziren Europar Batasuneko arrantza-flotak kudeatzeko. Kudeaketa plan hauek, epe luzerako helburuak eta horiek lortzeko neurriak zehazten dituzte. Kudeaketa plan batek helburuak betetzeko duen gaitasuna ebaluatzeko metodorik egokiena, simulazio-ikerketak dira. Kudeaketa estrategien ebaluazioa (Management Strategy Evaluation, MSE) kudeaketa estrategien errendimendua ebaluatzea ahalbidetzen duen simulazio-eredu bat da. Bertan, sistema-dinamiken inguruko zenbait hipotesi erabiliz, errealitate bat simulatzen da eta kudeaketa estrategien efektua simulatutako errealitate horretan baloratzen da kudeaketa estrategia indarrean sartu aurretik. MSEaren ezaugarri nagusietako bat da arrantza-floten kudeaketarako erabakietan ziurgabetasuna aintzat hartzen laguntzen duela.

Arrantza-flota mistoetan, espezie ugari harrapatzen dira aldi berean. Honela, harrapaketetan espezieak bereizteko ezintasunaren ondorioz, arrantza-flota mistoen kudeaketak, erronka gehigarria du kudeatzaileentzat, ezinezkoa baita espezie guztien kudeaketa helburuak aldi berean lortzea. Horrez gain, harrapatutako arrainak portura eramateko obligazioaren ondorioz, flota mistoek, arrantzatzeari utzi behar izaten diote espezie baten kuotara iristen direnean. Izan ere, ez dute espezie horren harrapaketak saihesteko aukerarik izaten beste espezie batzuk arrantzatzerakoan eta ondorioz, beste espezieen kuota bete gabe geratzen da.

Europar Batasunean, TACak ezartzea Europako Kontseiluaren ardura da. Europako Kontseiluak erabakiak hartzeko, besteak beste, ICES (International Council for the Exploration of the Sea)-ek harrapaketen inguruan emandako aholkuak hartzen ditu aintzat. ICESek, arrantza-aukeren inguruko kudeaketa aholkuak emateko, metodo eta harrapaketak ezartzeko arau (Harvest Control Rule, HCR) ezberdinak erabiltzen ditu espezie bakoitzarentzat eskuragarri dauden datuen eta informazioaren arabera. Ebaluazio kuantitatiboa duten espezieak datu-ugariko espezieak dira. Ebaluazio-eredu konbentzionalak aplikatu ahal izateko datu nahikorik agertzen ez duten espezieak, bestalde, datu-mugatuak dituzten espezieak dira.

Tesi honetan Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoak izan ditugu hizpide. Frantziako eta Espainiako ontziez osatuta dago gehienbat eta 150 espezietik gora harrapatzen dituzte. Harrapatzen dituzten espezie horietako laurdenak baino ez daude TAC eta kuota-sisteman sartuta, eta horietako zazpi eta lupia arrunta baino ez daude 2019an ezarri zen Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planean sartuta. TACak espezie bakoitzarentzat ezartzen dira espezie gehienen kasuan. Arraien eta zapoen kasuan, ordea, TACa genero-mailan zehazten da eta taldekako TACak aplikatzen dira, oso zaila baita genero berdineko espezieen artean bereiztea. Beste espezie guztiak inolako kuota-mugarik gabe harrapatzen dira. Horrek ezbaian jartzen du Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoak kudeatzeko martxan dagoen urte anitzeko kudeaketa planaren egokitasuna sistema osoaren jasangarritasuna bermatzeko.

2015ean, STECFk (Scientific, Technical and Economic Committee for Fisheries), Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planak Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen arrantza-jardueran eta ustiatutako espezieen jasangarritasunean izan dezakeen eragina ebaluatu zuen. Horretarako, bi simulazio-eredu aplikatu ziren: Frantziako flotan zentratua bata, eta Espainiakoan bestea, simulazioan 17 eta 12 espezie sartuz, hurrenez hurren. Espezie horietatik, baina, lehen simulazio-ereduan hiru espezie (legatza, mihi-arraina eta zigalak) eta bigarren simulazio-ereduan bi espezie (legatza eta oilarra) baino ez ziren dinamikoki simulatu. Beste espezie guztiak, esfortzu unitateko harrapaketa (Catch Per Unit Effort, CPUE) konstantearen irizpidea jarraituz sartu ziren simulazioan. Horren arabera, harrapaketek lotura lineala dute esfortzuarekin eta biomasarekiko independenteak dira. Tesi hau, STECFk Bizkaiko Golkoan jarduten duen arrantza-flota demertsal mistoen kudeaketa plana ebaluatzeko xedearekin 2015ean garatu zuen simulazio-ereduaren aplikazioa hobetzera bideratu zen. Aurreko lanarekin alderatuta, ikerketaren aplikazio eremua zabaldu genuen arrisku-egoera larrienean zeuden, gehien ustiatuta zeuden eta/edo flotarentzat ekonomikoki garrantzitsuenak ziren espezieak simulazioan sartuz, kontuan hartu gabe espezieen datu eskuragarritasuna. Simulazio-ereduan datu-ugariko espezieen populazio-dinamikak sartzeko, ICESek onartutako ebaluazio-eredu kuantitatiboetatik ateratako emaitzak erabili ziren, bestalde, datu-mugatuak dituzten lau espezie simulazioan sartzeko, espeziearen murrizteari buruzko analisia (Stock Reduction Analysis, SRA) erabili zen. Honek, arrantza kudeaketa estrategien errendimenduaren ebaluazio holistikoa egitea ahalbidetu zigun.

1.Kapituluan, tesiaren edukiak azaltzeko helburuarekin, CFPren bilakaera laburbildu zen, arreta, arrantza-flota mistoen kudeaketak dakartzan erronketan jarriz. Horrez gain, MSE esparrua aurkeztu zen, espezie anitzeko arrantza-flota mistoen MSEa egiteak ekar ditzakeen zailtasunak azalduz. Testuinguru horretan, simulazio-ereduan sartu behar diren espezieak aukeratzeko metodoak eta datu-mugatuak dituzten espezieen dinamikaren kalkulu fidagarriak lortzeko datu gutxi behar dituzten ebaluazio-ereduak aurkeztu ziren.

Arrantza-flota mistoek, espezie ugari harrapatzen dituzte eta ezinezkoa da espezie guztiak esplizituki simulazio-ereduan sartzea horrek suposatzen duen lan karga handiagatik, behar diren baliabide konputazionalengatik eta eskuragarri dauden datu mugatuengatik. Nolanahi ere, simulazioan sartutako espezieek, adierazgarritasun nahikoa bermatu behar dute flotaren jarduera deskribatzeko. Horrek bermatzen du kudeaketa estrategia batek jasankortasun biologikoan duen eragina zehatz ebaluatu ahal izatea. Ondorioz, espezie anitzeko simulazio-eredu bat aplikatzeko lehen pausoa eta erronka da, simulazioan sartu beharreko espezieak identifikatzea. Kasu askotan, espezieak aukeratzeko zio bakarra da lehorreratze guztiei edo arrantza-flotaren diru-sarrerei egiten dieten ekarpena; edo ekologikoki babestuak izateko beharra. Hala ere, espezieak sailkatzeko metodo bat ere badago ondoko alderdietan oinarrituz espezieak lehenestea proposatzen duena: lehorreratzeei egiten dieten ekarpena, batez besteko prezioa kiloko eta beren arrisku ekologikoa arrantzaren eraginarekiko. Proposamen hau, 2.Kapituluan egokitu eta aplikatu zen simulazioan sartu beharreko espezieak identifikatzeko. Honela, espezieak, gehien ustiatuta zeudenetatik, ekonomikoki garrantzitsuenak zirenetatik eta/edo arrisku ekologiko handienean zeudenetatik, beste muturreraino ordenatu ziren. Ustiatze maila kalkulatzeko, espezie bakoitzaren harrapaketek harrapaketa guztien barruan agertzen zuten ehunekoa hartu zen kontuan. Garrantzi ekonomikoa, bestalde, espeziearen kiloko batez besteko prezioa eta lehorreratutako kiloak biderkatuz kalkulatu zen. Espezieen arrisku ekologikoa kalkulatzeko, ebaluazio-eredu erdi-kuantitatibo bat erabili zen: produktibitate-suszeptibilitate analisia (Productivity-Susceptibility Analysis, PSA). PSAk, arrantza-flota misto baten xede diren espezieen jasangarritasun ekologikoa esparru bakar batean aztertzea ahalbidetzen du, espezieen artean datuen eskuragarritasunari dagokiola egon daitezkeen aldeak edozein direlarik ere. PSAn, arrisku potentziala zehazteko bi ezaugarri hartzen dira aintzat: produktibitatea (murrizketaren ostean espezieak aurreko egoerara bueltatzeko duen gaitasuna da) eta suszeptibilitatea (arrantza-jarduerek espeziearen jasangarritasunean duten eragin potentziala da). PSAren oinarria da, produktibitate baxuko eta suszeptibilitate altuko egoeretan, espeziea arrisku potentzial handian dagoela. Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoek suszeptibilitate ezberdineko aparailuak erabiltzen dituztenez, hasteko, espezieen arrisku potentziala aparailu bakoitzerako kalkulatu zen, era independentean; eta, ondoren, arrantza-jarduera guztiaren eragin metatua kalkulatu zen, suszeptibilitate metatuaren metodoa erabiliz. Arrisku potentzialaren ziurgabetasuna neurtzeko, datuen kalitatearen indize bat erabili zen. Indize horrek datu bakoitza puntuatzen du, datuaren kalitatearen arabera. Hiru faktoreetako bakoitzak (harrapaketak, diru-sarrerak eta arriskua) espezieak lehenesteko sailkapenari egindako ekarpena era orekatuan banatu zen. Horrez gain, diru-sarrerak eta harrapaketen bolumena kalkulatzeko aplikatu zen epealdiaren eragina ere aztertu zen, espezieak denboran zehar agertzen zuen posizioa alderatuz. Epealdi ezberdinetan, sailkapenen arteko aldeak oso txikiak izan ziren.

Espezie esanguratsuenak aukeratu eta gero, hurrengo pausua, populazio- eta flota-dinamiken informazioa epealdi historikorako eta proiekzio-epealdirako osatzea izan zen. Espezie-dinamikei dagokienez, datu-ugariko espezieek ebaluazio kuantitatiboa dute, eta beraz, haien populazio-dinamika simulazio-ereduan, ebaluazio-ereduko kalkuluetan oinarritu zen. Aukeratutako espezie gehienak baina, datu-mugatuak zituzten, eta horrenbestez, ez zegoen datu nahikorik ohiko ebaluazio-ereduak aplikatu eta emaitza hauetan oinarrituta hauek simulazio-ereduan sartzeko. Datu-mugatuak dituzten espezieak simulazioan sartzea, kudeaketa estrategien ebaluaketan dagoen erronka handienetako bat da oraindik. Simulazio-ikerketa askotan, datu-mugatuak dituzten espezieen dinamika, biologia-parametroetan eta ustiapen historikoaren inguruko hipotesietan oinarritzen da. Beste aukera bat da, datu gutxi behar dituen ebaluazio-eredu bat erabiltzea. SRA, datu gutxi behar dituen adinean oinarritutako ebaluazio-eredu bat da. SRA ereduak, harrapaketen eta abundantzia-indizearen inguruan eskuragarri dauden datuak eta bizi-historiari buruzko datuak konbinatzen ditu populazio-biomasaren eta ustiatze-mailaren kalkulu historikoak lortzeko. SRA eredua, hainbat simulazio ikerketetan erabili izan da datu-mugatuak dituzten espezieen populazio-dinamikak sartzeko. Ikertzaile ezberdinek SRA ereduaren errendimendua aztertu izan dute biologia-parametroetan alborapena eta behaketa- eta eredu-hipotesietan akatsak sartuz. 3.Kapituluan, SRA ereduaren errendimenduan sakondu zen. Zehazki, eta "auto-proba" simulazio bitartez, SRA ereduak espeziebiomasa eta ustiatze-maila historikoaren kalkuluak lortzeko duen gaitasuna aztertu zen datuen eskuragarritasunari, populazioaren ustiatze mailari, hasierako populazio mailari eta arrantza-jardueraren selektibitatearen zehaztasunari zegozkien agertoki ezberdinetan. Honi esker, SRA eredua datu-mugatuak dituzten espezieen dinamika era fidagarrian zehazteko zein kasutan aplika daitekeen identifikatu zen. Oro har, eta espero bezala, zenbat eta datu gehiago eta zenbat eta epealdi luzeagoa aintzat hartu, are eta zehatzagoak ziren SRA eredutik lortutako kalkuluak. Eskuragarri zeuden datuak arrantzarik gabeko egoera batetik abiatzen zirenean, espeziearen belaunaldi bateko luzera zuten guztizko harrapaketa eta abundantzia-indizea erabiltzea nahikoa zen populazio- eta ustiatze-mailen inguruko kalkulu zehatzak lortzeko. Datuak ustiatze-egoera batetik hasten zirenean aldiz, bi belaunaldi gutxienez barnean hartzen zituzten guztizko harrapaketak eta abundantzia-indizea behar ziren ebaluazioeredutik kalkulu zehatzak lortzeko. Epealdi horren luzera bi belaunalditik beherakoa zenean, espeziearen belaunaldi bateko luzera zuen datu osagarriak behar ziren kalkulu zehatzak lortzeko. Harrapaketen batez besteko luzeraren datuak nahikoak ziren kalkulu erlatibo zehatzak (murriztea, adibidez) lortzeko. Arrantzatu gabeko errekrutatzeari buruzko kalkulu zehatzak lortzeko, ordea, luzeraren edo adinaren araberako datuak behar ziren. Horrez gain, ikusi genuen espeziearen hazkunde somatikoaren aldakortasuna baxua zenean eta biologia-parametroak ezagutzen zirenean, luzeraren araberako datuen edo adinaren araberako datuen erabilerak antzeko kalkuluak ematen zituela. Kasu guztietan, espeziearen belaunaldi bat aintzat hartzen zuen adinaren araberako abundantzia-indizea eta luzeraren edo adinaren araberako harrapaketa datuak, harrapaketa guztien datuei eta abundantzia-indizeari buruzko datuei gehitzea, kalkulu zehatzak lortzeko aukerarik aproposena zen, arrantza-jardueraren selektibitatea ezezaguna zenean bereziki.

4.Kapituluan, FLBEIA (Bio-Economic Impact Assessment in FLR) simulazio-eredua aplikatu zen. 2.Kapituluan lortutako emaitzak oinarri hartuta, 28 espezie sartu ziren simulazioan: horietako 13 adinean oinarritutako populazio-dinamikak erabiliz eta besteak CPUE konstantearen ikuspegia erabiliz. Adinean oinarritutako populazioei dagokienez, 13tik bederatzi datu-ugariko espezieak ziren eta beren populazio-dinamika simulazioan, beren ebaluazio-ereduko kalkuluetan oinarritu zen. Gainerako lau espezieak aldiz, katuarraina, haitzetako barbarina, arraia gastaka eta arraia zerra, datumugatuak zituzten espezieak ziren eta beren populazio-dinamikak simulazioan, ondorioz, SRA eredua oinarri hartuta sartu ziren. Izan ere, 3.Kapituluko emaitzen arabera, espezie hauen datu eskuragarritasuna nahikoa zen SRA eredua aplikatu eta beren populazio-dinamikari buruzko kalkulu zehatzak lortzeko. 2015ean egindako simulazioarekin alderatuta, legatzaz, mihi-arrainaz eta oilarraz aparte, datu-ugariko beste 6 espezie (zapo beltza, zapo zuria, txitxarro beltza, berdela, lupia arrunta eta bakalada) eta datu-mugatuko 4 espezie (katuarraina, haitzetako barbarina, arraia gastaka eta arraia zerra) dinamikoki sartu ziren simulazioan. Aurreko lanean zigalaren populaziodinamika sartu zen baina simulazio-ikerketa honetan ez zen zigalaren populazioa dinamikoki sartu, 2016tik aurrera bere ebaluazioa ur-azpiko TV ikerketa-indizean oinarritzen delako eta horren emaitzak ezin direlako ohiko populazio-dinamika eredu batera irauli. Tesi honetan, ondorioz, CPUE konstantearen ikuspegia erabili zen zigala simulazioan sartzeko. Horrez gain, 79 modalitatetan banatutako 51 flota sartu ziren simulazioan, ICESek Bizkaiko Golkoan jarduten duen arrantza-flota demersal mistoen kontsiderazioa eraikitzeko erabiltzen duen floten segmentazio berdina erabiliz. Ondokoak sartu ziren simulazioan: 16 flota eta 39 modalitate-konbinazio Frantziako ontzientzat, 7 flota eta 12 modalitate-konbinazio Espainiako ontzientzat, eta modalitate bana zuten 28 flota, flota bakoitzak espezie populazio bat arrantzatzen zuelarik, Bizkaiko Golkotik kanpo gertatzen ziren harrapaketak aintzat hartzeko. Simulatutako errealitate horretan, egungo urte anitzeko kudeaketa planak eta beste kudeaketa estrategia batzuk arrantza-jardueran eta ustiatutako espezieen jasangarritasunean zuten eragina ebaluatu zen, harrapatutako arrainak portura eramateko obligazioarekin eta gabe. Ikusi genuen, egungo kudeaketa planak datu-ugariko espezie guztien eta datu-mugatuak zituzten lau espezieetako bakar baten, hau da, katuarrainaren, jasangarritasuna bermatzen zuela. Haitzetako barbarinaren, arraia gastakaren eta arraia zerraren biomasak, aitzitik, erreferentziazko biomasa mugatik behera egoteko aukera handiak agertzen zituen epe luzera, harrapatutako arrainak portura eramateko obligazioa bete ala ez. Haitzetako barbarina ez dago TAC eta kuota-sisteman eta arraia gastaka eta arraia zerra kudeatzeko aholkuak emateko abundantzia-indizean oinarritzen den HCRa erabiltzen da. Ikusi genuenez, haitzetako barbarinaren eta arraia gastakaren jasangarritasuna dezente hobetu zen kudeaketan zenbait aldaketa egin eta gero. Zehazki, haitzetako barbarina TAC eta kuota-sisteman sartu zenean eta bere kudeaketa aholkua abundantzia-indizean edo MSYan oinarrituta eman zenean; eta arraia gastakaren kudeaketa aholkua emateko, abundantzia-indizean oinarritutako HCRa erabili beharrean MSY aholku-araua erabili zenean. Arraia zerraren egoerak, aitzitik, ez zuen hobera egin aztertutako kudeaketa estrategia berrirekin. Hau simulazioaren lehen urtean arraia zerrak zuen biomasa maila baxuari egotzi zitzaion. Simulazioaren lehen urtean, arraia zerraren biomasa erreferentziazko biomasa mugatik behera egoteko probabilitatea %50etik gorakoa zen. Populazio demertsalen ezaugarri nagusiak, arraia zerra barne, produktibitate baxua, bizi-ziklo luzea eta heldutasun berantiarra dira, eta honek, populazioa murrizten den egoera batean berreskuratzeko gaitasun baxua ematen die. Ondorioz, arraia zerraren populazioa berreskuratzeko, azterlan honetan aztertu ditugunak baino kudeaketa estrategia zehatzagoak behar dira. Ikusi genuen, baita ere, arraientzat eta zapoentzat ezarritako taldekako TAC neurriek eragina zutela arrantza-jardueran eta espezie populazioen jasangarritasunean. Alde batetik, taldekako TAC neurriek arrantza-jardueraren egonkortasuna sustatzen zuten. Hala ere, honek, espezie populazioen epe luzeko jasangarritasuna zalantzan jartzen zuen. Zehazki, TACak arraia eta zapo espezie bakoitzarentzat zehaztu zirenean, arraia gastakaren TACak flota gehienen arrantza-jarduera mugatu zuen eta horrek beste espezieen arrantza-aukerez baliatzeko ezintasuna ekarri zuen. Taldekatutako TACak ezartzeak muga horiek arindu zituen, baina espezie bakoitzari zegokion harrapaketa-mailak gainditzea ahalbidetu zuen, eta horrek eragin negatiboa izan zuen populazioen biomasa-mailetan.

Tesi honetan lorturiko emaitzek, Bizkaiko Golkoan jarduten duten arrantza-flota demersal mistoen kudeaketarako erabiltzen diren neurrien ebaluazio holistiko bat egitea ahalbidetu zuten. Praktikan, emaitza hauek kudeaketari loturiko aholkuak hobetzeko erabil daitezke eta CFPren xede nagusia (harrapatutako espezie guztien jasangarritasuna bermatzea eta, aldi berean, arrantza-jarduera epe luzera mantentzea) lortzeko bidean aurrera egiten lagundu dezakete.

Summary

We are in the sixth and potentially most extreme biodiversity crisis and with it, the future of human health is endangered. Overfishing is one of the main sources of marine biodiversity loss, so sustainable management strategies are crucial for recovering or maintaining the marine ecosystem's biodiversity. Since 1970 the management of European Union fishery resources has been operationalized through the Common Fisheries Policy (CFP). In the latest CFP reform in 2013, the overarching management objective was set at the Maximum Sustainable Yield (MSY), which means to catch the largest amount of fish over time while keeping the stock at the maximum production level. The management of all the stocks should also comply with the precautionary principle according to which the lack of full scientific certainty should not be the reason for postponing the implementation of measures that could prevent environmental degradation. Fisheries are managed by a variety of management tools, from Total Allowable Catches (TACs) to technical measures such as minimum landing size, mesh size limitation or marine protected areas. An important element of the last CFP reform is the implementation of the landing obligation whose aim is to eliminate discards by avoiding and reducing accidental or unwanted catches. According to this regulation, all the stocks subject to a catch limit and/or a legal minimum conservation reference size must be landed, when caught. In addition, with the last reform of the CFP, the multiannual management plans became one of the most relevant tools for fisheries management in the European Union. These plans establish long-term management goals for the stocks and the corresponding measures to achieve them. The most appropriate method for evaluating the effectiveness of a given management plan in achieving its objectives is through simulation studies. The Management Strategy Evaluation (MSE) is a simulation framework that allows to evaluate the performance of management strategies under different hypotheses about the system dynamics in a simulated reality before the management strategy is put in place. One of the main characteristic of the MSE is that it formalizes the incorporation of uncertainty in the fisheries management decision making process.

In mixed-fisheries, several stock are caught simultaneously. The management of mixed-fisheries implies an extra challenge for the managers as the impossibility of discriminating between stocks makes it impossible in practice to reach the single stock management objectives at the same time. In addition, under the landing obligation, the mixed-fisheries activity must be stopped when the quota of a stock is reached if the fishers cannot avoid fishing that stock when fishing for the other stocks. Thus, in this situation, the quota of the rest of the stocks might be underutilized.

In the European Union, the setting of TACs is the responsibility of the European Council, that among other scientific bodies bases the decisions on the catch advice provided by ICES (International Council for the Exploration of the Sea). ICES uses different methods and harvest control rules (HCRs) to provide management advice on fishing opportunities based on the data and knowledge available for each stock. Stocks

with quantitative assessment are data-rich stocks and stocks for which available data are not enough to apply conventional assessment models are data-limited stocks.

In this thesis we focused on the demersal mixed-fisheries operating in the Bay of Biscay, which primarily involved French and Spanish vessels and encompass the capture of over 150 different species. Among these species, only around one-fourth of them are included in the TAC and quota system, and only seven of them and sea bass are included in the multiannual management plan for Western Waters and adjacent waters, which was implemented in 2019. The TACs are given at single stock level for most of the stocks. However, for rays and anglerfishes they are given at genus level using grouped TACs due to the difficulty to distinguish between stocks of the same genus. The rest of the stocks are caught without any quota restriction. This raises the question of whether the current multiannual management plan of the demersal mixed-fisheries operating in the Bay of Biscay is sufficient to ensure the sustainability of the whole system.

In 2015 the STECF (Scientific, Technical and Economic Committee for Fisheries) evaluated the potential impact of the Western Waters and adjacent waters multiannual management plan on the activity of the demersal mixed-fisheries operating in the Bay of Biscay and on the sustainability of the exploited stocks. Two simulation models were implemented: one focused on the French fleet, and another on the Spanish fleet, including 17 and 12 species respectively. However, the population dynamics of only three of them (hake, sole and nephrops) in the former and two of them (hake and megrim) in the latter were simulated dynamically. The other species were included in the simulation using a constant Catch Per Unit Effort (CPUE) approach, whereby the catch is linearly related to effort and is independent of the biomass. The research undertaken in this thesis was focused on the improvement of the simulation implemented in 2015 by STECF for the evaluation of the multiannual management plan of the demersal mixed-fisheries in the Bay of Biscay. In comparison to previous work, we broadened the scope of the work including in the simulation those stocks most at risk, most exploited and/or economically most important for the fishery regardless of the stocks data availability. Quantitative stocks assessment models approved by ICES were used to condition the population dynamics of the data-rich stocks, and the Stock Reduction Analysis (SRA) was applied to four data-limited stocks. This allowed us to provide a holistic evaluation of the performance of management strategies for the fishery.

In Chapter 1, to familiarise the reader with the contents of the thesis, the evolution of the CFP was summarized, focusing on the challenges faced by mixed-fisheries management. In addition, the MSE framework was introduced highlighting some of the difficulties to condition multi-stock MSE models for mixed-fisheries. In that context, methods to select the stocks to be included in a simulation model and low data demand assessment models to get reliable representation of the dynamics of data-limited stocks were presented.

Mixed-fisheries catch a large number of species, making it practically impossible to include all of them in the simulation explicitly. This is primarily due to the significant workload, computational resources needed, and limitations in available data. Nonetheless, the stocks included in the simulation should be representative enough to describe the activity of the fishery. This ensures that the impact of a management strategy on biological sustainability can be accurately assessed. Thus, the first step and challenge to implement a multi-stock simulation model is to identify the stocks to be included in the simulation. In many cases, the stocks are selected based exclusively on their contribution to the total landings and income to the fishery, or on the need to be ecologically protected. However, there is an approach to rank stocks according to their contribution to landings, average price per kilogram, and ecological risk for the effects of fishing. This approach was modified and applied in Chapter 2 to identify the stocks that should be included in the simulation. The species were ranked from those most exploited, economically most important and/or most at risk for the effects of fishing activity, to the least. The exploitation was measured in terms of the contribution of the stock to the catches and the economical importance was calculated by multiplying the mean price per kilogram and the landed kilograms of the stock. The ecological risk was assessed using a semi-quantitative risk assessment model called Productivity-Susceptibility Analysis (PSA). The PSA allowed to assess within the same framework the ecological sustainability of a large number of stocks involved in a mixed-fishery despite the differences in data availability. In the PSA, the potential risk is derived from two characteristics: the productivity, which determines the capacity of a stock to recover from depletion, and the susceptibility, which quantifies the potential impact of the fishing activities on the stock. The basis of the PSA is that lower productivity and higher susceptibility imply higher potential risk for the stock. As the demersal mixed-fisheries operating in the Bay of Biscay are composed by different gears with different susceptibilities, first, we calculated the potential risk of stocks to each gear independently; then, we calculated the cumulative impact of overlapping fishing activities using the aggregated susceptibility method. The uncertainty of the potential risk was measured using a data quality index that scored each data based on the quality of the data used. The contribution to the ranking of each of the three factors (catch, revenue and risk) was balanced equally. In addition, the sensitivity of the ranking to the time period used for revenue and catch volume calculation was analysed comparing the stock's position along time, and the differences were low.

Once the most relevant stocks were selected, the next step was to condition the population and fleet dynamics in the historic and projection period. Regarding the stock dynamics, data-rich stocks have a quantitative assessment so their population dynamics were conditioned in the simulation model based on their assessment model estimates. However, most of the stocks selected were data-limited stocks, lacking sufficient data to apply conventional assessment models. Thus, including them in the simulation remained one of the major challenges in the evaluation of management strategies for data-limited stocks. In many simulation studies, data-limited stocks have been conditioned based on life-history traits and assumptions about the historical exploitation. Alternatively, low data demand assessment models have also been used. SRA is a low data demand age-structured assessment model that combines the available catch and abundance index data with life-history parameters to obtain the historical estimates of population biomass and exploitation levels of a stock. The SRA model has been used in several simulation studies to define the population dynamics of data-limited stocks. Some authors have evaluated the performance of the SRA model under bias in biological parameters and error in observation and model assumptions. In Chapter 3 we made progress in understanding the performance of the SRA model. Specifically, the ability of the SRA model to predict estimates of historical stock biomass and exploitation levels under alternative data availability scenarios, population exploitation levels, initial population assumptions and misspecified fishery sensitivity was tested by "self-test" simulation. This allowed us to identify in which cases SRA could be applied to condition data-limited stock population dynamics reliably. In general, as expected, the more data and the longer time-series used, the more accurate the SRA model estimates. When the available data started from an unfished condition, time-series of total catch and abundance index data that covered one stock generation time was sufficient to obtain good enough population and exploitation level estimates. When the data started from an exploited condition, the time-series of total catch and abundance index data had to cover at least two stock generation times to obtain accurate estimates. When the length of the data time-series was shorter than two stock generation times, additional data covering a stock's generation time was needed. Mean length in the catch data may be sufficient to obtain accurate relative estimates (e.g., depletion). However, for accurate estimates of unfished recruitment, age- or length-structured data were needed. In addition, we found that when the variability in the stock's somatic growth was low and the biological parameters were known, the use of length-structured data or age-structured data gave similar estimates. In all cases, the addition of one generation time of a stock's index-at-age data and age- or length-structured catch data to total catch and abundance index data was the best option to obtain accurate estimates, especially when the fishery selectivity was unknown.

In Chapter 4, the simulation model was implemented using FLBEIA (Bio-Economic Impact Assessment in FLR). Based on the results from Chapter 2, 28 stocks were included in the simulation: 13 were included using age-structured population dynamics, and the rest were included using a constant CPUE approach. Regarding the age-structured stocks, nine out of 13 were data-rich stocks so their conditioning in the simulation was based on their assessment model estimates, whereas the remaining four stocks, catshark, red mullet, thornback ray and cuckoo ray, were data-limited stocks and their conditioning in the simulation was based on the SRA model fit. In fact, according to the results in Chapter 3, the data availability was enough to obtain accurate estimates of their population dynamics from the SRA model. Comparing to the simulation done in 2015, apart from hake, sole and megrim, another 6 data-rich stocks (black anglerfish, white anglerfish, horse mackerel, mackerel, sea bass, and blue whiting) and 4 data-limited stocks (catshark, red mullet, thornback ray and cuckoo ray) were included dynamically in the simulation. The population dynamics of Norway lobster was incorporated in the previous work however it was not included in this simulation study because since 2016 its assessment is based on an underwater TV survey whose results were not translatable to a classical population dynamics model. Thus, in this thesis, a constant CPUE approach was used to include Norway lobster in the simulation. In addition, 51 fleets divided into 79 metiers were included in the algorithm with the same segmentation as used in the provision of ICES mixed-fisheries considerations for the Bay of Biscay. Sixteen fleet and 39 metier combinations for French vessels, 7 fleet and 12 metier combinations for Spanish vessels, and 28 fleet each with one metier to account for the catches of each stock that occur outside the Bay of Biscay were included in the simulation. In this simulated reality, the impact of current multiannual management plan and alternative management strategies on the fishery activity and on the sustainability of the exploited stocks was evaluated considering the implementation or non-implementation of the landing obligation. We found that the current management plan ensures the sustainability of all the data-rich stocks and one of the four data-limited stocks included in the simulation, namely, catshark. On the contrary, the biomass of red mullet, thornback ray and cuckoo ray had a high probability of being below the limit reference biomass in the long-term, regardless of whether the landing obligation was fully implemented or not. Red mullet is not included in the TAC and quota system and the management advice of thornback ray and cuckoo ray is given using the HCR based on an abundance index. We found that the sustainability of red mullet and thornback ray improved substantially with some changes in their management. Specifically, when red mullet was included in the TAC and quota system and its advice was given based on a HCR that uses an abundance index or when the MSY advice rule was used, and when the HCR of thornback ray was upgraded from the HCR based on an abundance index to the MSY advice rule. However, the status of cuckoo ray did not improve with any of the alternative management strategies tested. This was due to the low biomass level of cuckoo ray in the initial year of the simulation. There was a probability of over 50% that the biomass of cuckoo ray would fall below the limit reference biomass in the first year of the simulation. Demersal stocks, including cuckoo ray, are characterized by low productivity, long lifespan, and late maturity, resulting in a reduced capacity to recover from low biomass levels. Consequently, the recovery of this stock requires a more specific management strategy than the ones analysed in this study. We also found that the implementation of grouped TACs for rays and anglerfishes had mixed effects on the fishing activity and stock sustainability. On one hand, the grouped TACs contributed to the stability of fishing operations. However, they also raised concerns regarding the long-term sustainability of the stocks. Specifically, when individual TACs for rays and anglerfishes were utilized, the TAC for thornback ray constrained the fishing activity of most of the fleets, resulting in misuse of other stocks fishing opportunities. The implementation of grouped TACs alleviated these limitations but led to fishing levels surpassing the recommended thresholds at the single-stock level, negatively impacting the biomass levels of the stocks.

The results obtained in this thesis contributed to providing a holistic evaluation of the management of the demersal mixed-fisheries operating in the Bay of Biscay. In practice, it could lead to an improvement of the management advice and help moving forward to overarching the main objective of the CFP which is to ensure the sustainability of all the stocks caught while maintaining the fishing activity in the long-term.

1 Kapitulua

Sarrera Orokorra

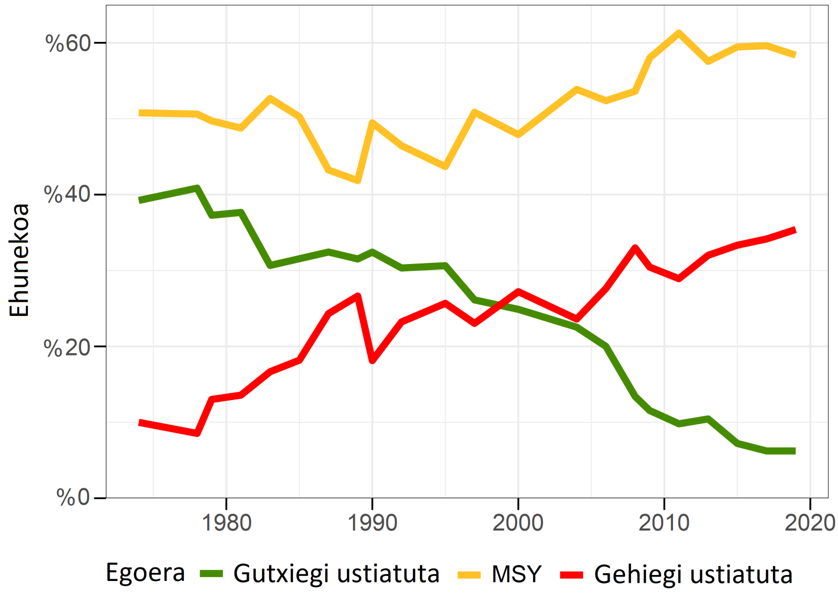
The English version of this chapter is in Appendix A

Jane Goodallek ondokoa dio *We are nature* izeneko dokumentalean: "we may think that nature is something separate and we can distance ourselves and live in our little bubble, but it's not true". Gizakiaren osasuna, fisikoa zein mentala, ekosistemen biodibertsitateari lotua dago (Naeem et al., 2016). Biodibertsitateak ezinbesteko zerbitzuak eskaintzen dizkio gizartearen ongizateari (Naeem et al., 2016). Esaterako, ura eta elikagaiak eskaintzen ditu, energia garbia, farmazia- eta kosmetika-produktuak, itsasertzeko babesa eta klimaren erregulazioa, jarduera ekonomikoetan zeharkako paper garrantzitsua duela ahaztu gabe (Barbier, 2017; Hanley et al., 2015). Espezie guztiek dute beren eginkizuna ekosisteman eta espezie txiki eta itxuraz hutsal baten desagerpenak domino-efektua izan dezake beste espezie batzuengan, ekosistema kolapsatuz eta, ondorioz, gizakien osasuna arriskuan jarriz (Naeem et al., 2016).

Adituen ustez seigarren biodibertsitate-krisian murgilduta gaude, eta hau, potentzialki muturrekoena izan daiteke (Banks-Leite et al., 2020). Aurreikuspenen arabera, espezie asko inoiz baino azkarrago desagertuko dira (Banks-Leite et al., 2020). Itsasoak ez dira salbuespena, eta itsasoko ekosistemak abiadura azkarrean eta maila kezkagarrian desagertzen ari dira mundu osoan zehar (Barbier, 2017). Mundu osoan ematen ari den itsas ekosistemen eta biodibertsitatean degradazio orokorra, giza-jardueren eragin metatuaren ondorio da (McQuatters-Gollop et al., 2022). Mehatxu garrantzitsuenetako batzuk, gizakien presio zuzenaren ondorio dira: itsasertzeko habitaten suntsiketa, kutsadura, bertakoak ez diren espezieen sartzea eta arrantza (McQuatters-Gollop et al., 2022). Baina giza-jardueraren zeharkako eraginak ere badaude, klima-aldaketa tartean (McQuatters-Gollop et al., 2022). Gizateriaren hedatzeak eta hazkundeak biodibertsitatearen galera areagotzen du eta lotura positiboa dauka talde taxonomikoen eta ekosistema-egituren gainbeherarekin (Lotze et al., 2011). Egun, giza-eskaera sostengatzeko itsasoek duten gaitasuna bidegurutzean dago (Duarte et al., 2020).

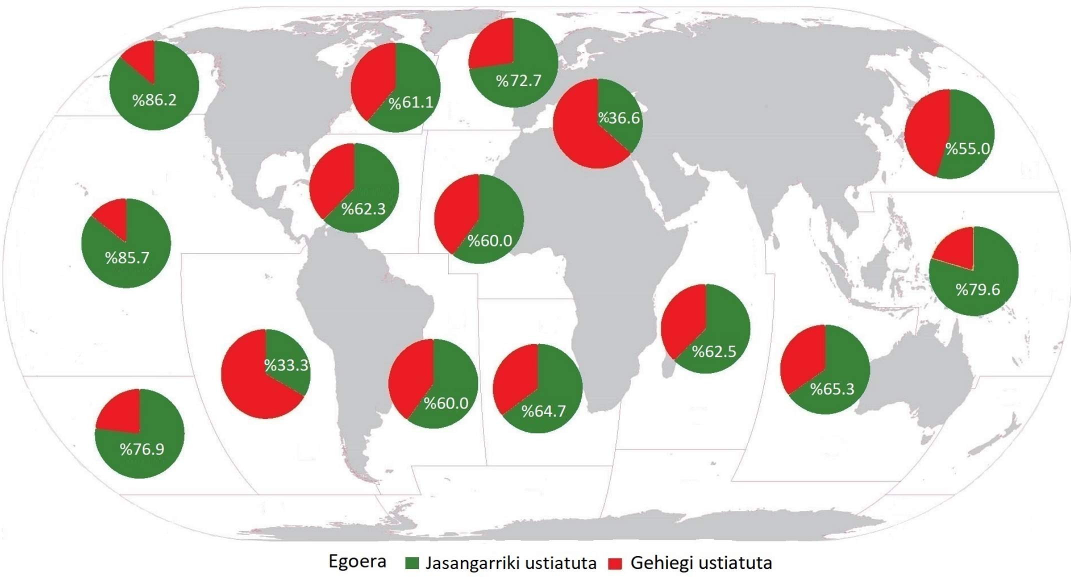
XIX.mende amaiera arte, uste orokorra zen itsasoak hain handiak zirenez, arrantza-jarduerek ezin zutela inolako eragin negatiborik izan arrain-espezieen abundantzian. Thomas Henry Huxleyk ondokoa adierazi zuen 1883ko Nazioarteko Arrantza Erakusketaren irekiera-ekitaldian: "the cod fishery, the herring fishery, the pilchard fishery, the mackerel fishery and probably all the great sea fisheries, are inexhaustible; that is to say, that nothing we do seriously affects the number of the fish. And any attempt to regulate these fisheries seems consequently, from the nature of the case, to be useless" (Huxley, 2020). XIX. mendea amaitu aurretik, ordea, munduko kala ezberdinetan arrainen abundantzia beherantz zihoala ikusita, aditu batzuk populazioen ustezko iraungiezintasun horren aurka agertu ziren eta giza-jarduerak itsasoko baliabideak suntsitzeko gaitasuna zuela aldarrikatzen hasi ziren (Sims and Southward, 2006). Egun, aski ezaguna da gehiegizko arrantza-jarduera itsasoko baliabideen gainbeheraren arrazoi nagusietako bat dela (Barbier, 2017).

Nazio Batuetako Elikadura eta Nekazaritza Erakundearen (Food and Agriculture Organization, FAO) arabera, gehiegi arrantzatuta dauden itsasoko populazioen kopuruak gora egin du azken hamarkadetan (1.1.Irudia), 2019an arrantza-baliabide guztien %35eraino iritsiz (FAO, 2022). Biologikoki jasangarriak diren espezieen artean (hau da, gehiegi arrantzatuta ez dauden horien artean), errendimendu jasangarri maximoan (Maximum Sustainable Yield, MSY) arrantzatzen diren espezieen ehunekoak %51tik %58ra jauzi egin du 1974 eta 2019 bitartean eta gutxi arrantzatzen diren espezieen ehunekoak, ordea, behera egin du: %39 1974an eta %6 2019an (1.1.Irudia). Lehorreratzeetan erreparatuz, eta kalkuluen arabera, gehiegi ustiatuta dauden espezieen lehorreratzeek lehorreratze guztien %18 suposatzen dute (FAO, 2022).



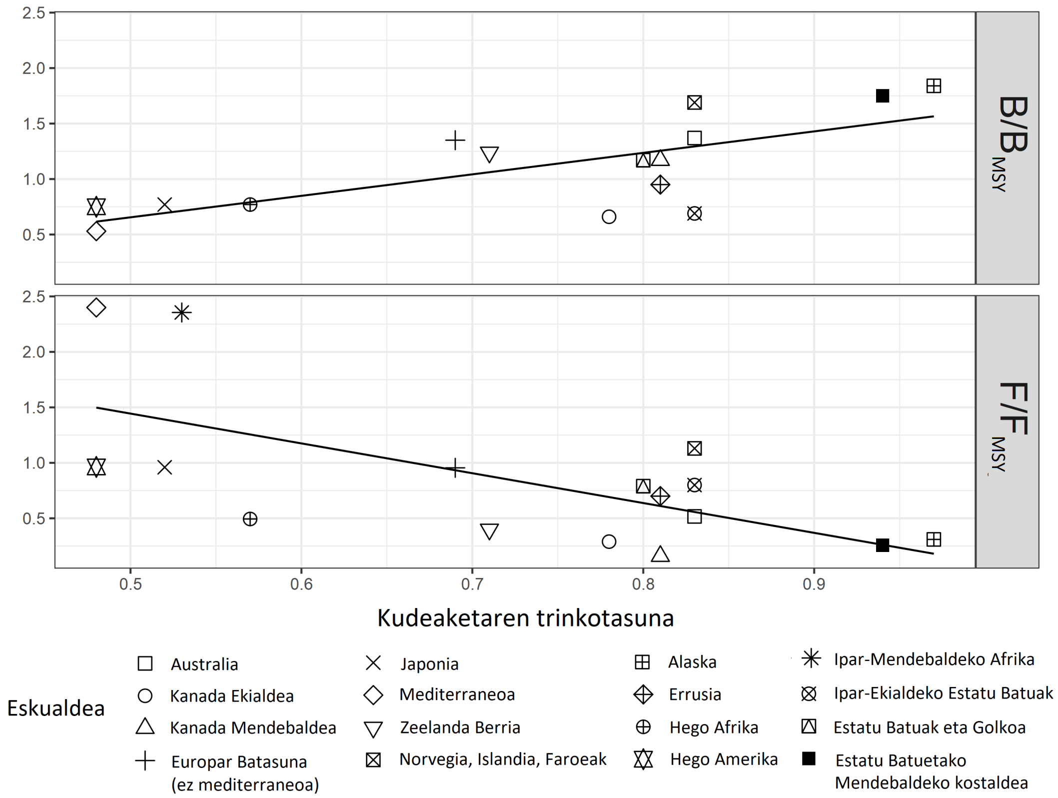
Irudia 1.1: Joera globalak mundu osoko arrain-espezieen egoerari dagokionez, 1974-2019. FAO (2022)ren datuetatik egokituta.

Gehiegi ustiatuta dauden espezieen ehunekoa eskualdearen arabera aldatzen da (FAO, 2022). 2019an, gehiegi ustiatuta zeuden espezieen ehunekorik handiena Pazifikoko Hego-ekialdean zegoen (%66.7) (1.2.Irudia). Horren atzetik Mediterraneoa eta Itsaso Beltza zeuden (%63.4) (1.2.Irudia). Aitzitik, gehiegi ustiatutako espezie gutxien zituzten eskualdeak honako hauek ziren: Pazifikoko Ipar-ekialdea, Pazifikoko Ekialde-erdialdea, Pazifikoko Mendebalde-erdialdea eta Pazifikoko Hego-mendebaldea (%13 eta %23 bitartean) (1.2.Irudia). Beste eskualde batzuetan, gehiegi ustiatuta zeuden espezieen ehunekoa %27 eta %45 artean zegoen (1.2.Irudia).



Irudia 1.2: Jasangarriki eta gehiegi ustiatutako espezieen ehunekoak FAOk zehaztutako kaletan, 2019an. Zenbakiek biologikoki jasangarriak ziren espezieen proportzioa adierazten dute. FAO (2022)ren datuetatik egokituta.

Eskualde bakoitzean gehiegi ustiatuta dauden espezie populazioen ehunekoak erlazio zuzena erakusten du zientzia oinarri hartuta eskualde horretan aplikatzen diren arrantza-jarduera kudeatzeko estrategien trinkotasunarekiko (Hilborn et al., 2020). Arrantza-jardueraren kudeaketa trinkoa egiten duten eskualdeetan, batez beste, espezieen abundantzia hobetzen ari da, edo beren abundantzia kudeaketa helburuetan mantentzen da bederen (1.3.Irudia). Kudeaketa hain zorrotza egiten ez duten eskualdeetan, aitzitik, espezieen egoera kaskarragoa da eta harrapaketa-mailak askoz altuagoak dira (1.3.Irudia). Horrek erakusten du arrantza-jarduera zientzian oinarrituta kudeatzeak, berebiziko garrantzia duela itsasoko ekosistemak berroneratzeko edo, behintzat, egoera osasuntsuan mantentzeko (Hilborn et al., 2020).



Irudia 1.3: Eskualdeko biomasaren batez besteko geometrikoa zati MSY biomasa (B/B_{MSY}, goialdean) eta harrapaketen heriotza-tasa zati MSY harrapaketen heriotza-tasa (F/F_{MSY}, behealdean), bertan egiten den kudeaketaren trinkotasunari dagokionez. Marra jarraituek datuen egokitzapen lineala adierazten dute. Hilborn et al. (2020)en lanetik egokituta.

1.1 Arrantza-floten kudeatzea Europar Batasunean

1.1.1 Arrantza Politika Erkidea

Arrantza politika erkidea (Common Fisheries Policy, CFP) Europar Batasuneko arrantza kudeatzeko xedez zehaztutako esparrua da. CFPak, arrantza-jarduera eta akuikultura, ingurumen-, ekonomia- eta gizarte-mailan jasangarriak direla eta hauek Europar Batasuneko hiritarrentzat elikadura osasuntsuaren iturri direla bermatzeko helburua duten arauak biltzen ditu (Casey et al., 2016).

CFPren jatorria da Europako Batzordeak 1967.urtean argitaratu zuen “Basic principles for a Common Fisheries Policy” izeneko txostena (Penas, 2016). 1970ean, Europako Kontseiluak lehen aldiz onartu zuen arrantza-produktuen merkatua antolatzeko egitura komuna ezartzea xede zuen legedi berezia eta arrantza-floten kudeaketarako egiturazko politika bat aplikatzen hasi zen (Penas, 2016). Garai hartan, arrain-populazio garrantzitsuenak egoera osasuntsuan zeudela uste zen eta arrantza-jarduera kudeatzeko metodologien garapena mugatua zen oraindik (Penas, 2016). Honela, legediaren jomuga baliabideen kontserbazioan jarri ordez, arrantza-baliabideen merkatuan jarri zen (Penas, 2016).

CFP 1983.urtean jaio zen, 171/83 (EEK) Erregulazioa eta 170/83 (EEK) Erregulazioa onartzearekin batera. Lehenak, neurri tekniko ezberdinak zehaztu zituen arrantza-baliabideen kontserbazioa bermatzeko: sare-begien tamaina mugak, nahi gabeko harrapaketen kuotak, lehorreratzen diren banakoen tamaina minimoak eta babestutako guneak, besteak beste. Bigarrenak, bere aldetik, arrantza-baliabideak kontserbatzeko eta kudeatzeko araubide orokorra ezarri zuen, araubideko neurririk garrantzitsuena baimenduta dagoen guztizko harrapaketa (Total Allowable Catch, TAC) zelarik (Penas, 2016). TACek adierazten dute arrantza-flota batek edo arrantza-eremu batean epe jakin batean harrapa daitekeen gehienezko arrai edo beste itsas espezieren gehiengo kopurua. TACak Europar Batasuneko estatu kideen artean partekatzen dira arrantza-kuotak gisa eta estatu kide horiek beren arrantza-kuotak flota ezberdinen artean banatzen dituzte. 1983tik, arrantza-kuotak egonkortasun erlatiboaren printzipioari jarraiki partekatzen dira. Bere izenak adierazten duen bezalaxe, esan nahi duena da arrain-espezie bakoitzaren TACa proportzio finko baten arabera banatzen dela Europar Batasuneko estatu kideen artean (Hoefnagel et al., 2015; Penas, 2016). Garai hartan, nahiz eta zenbait espezieren kudeaketa zehaztuta egon harrapa zitekeen arrain-bolumena mugatuz edo/eta neurri tekniko ezberdinak ezarriz, oro har, arrantza-jarduerek sarbide askearen printzipioari jarraitzen zioten (Penas, 2016).

Hasiera-hasieratik, CFP hamar urtero berrikusi eta aldatzen da, gutxi gorabehera. 1992ko berrikuspenak flota-gaitasunaren eta harrapaketa potentzialen arteko desorekari aurre egin behar izan zion, 1985an Groenlandia Europar Batasunetik irten, 1986an Espainia eta Portugal sartu eta 1990an Alemania berriro batu eta gero. Horrez gain, lizentzia-sistema bat ezarri zen, kala ezberdinetara sartzeko eskubidea izango zuten flotak kudeatzeko. Ezarritako neurriak, baina, ez ziren behar bezain eraginkorrak izan eta arrain-espezie askoren gainbehera are azkarragoa izaten jarraitu zuen. Horrek, 2002.urteko CFPren erreforma ekarri zuen (Breuer, 2022).

2002ko CFPren erreformarekin, 2371/2002 (EK) Erregulazioa ezarri zen eta lehen aldiz arrantza-kudeaketaren xedea arrantza-baliabideen erabilera jasangarria ziurtatzean oinarritu zen, arrantzaleei eta kontsumitzaileei enplegu eta ordainsari egonkorrak bermatuz (Breuer, 2022). 2002ko berrikuspenak, jasangarritasuna zientzialarien aholkuetan eta zuhurtasunaren printzipioan oinarritu beharra zegoela adierazi zuen. Zuhurtasunaren printzipioari jarraiki, erabateko ziurtasun zientifiko ezak ez luke ingurumen-narriatzea geldi dezaketen kudeaketa neurrien ezarpena atzeratzeko arrazoia izan behar (UNCED, 1992). Horrez gain, CFPren 2002ko berrikuspenak epe luzerako ikuspegia aplikatu zuen arrantza-floten kudeaketarako: urte anitzeko berreskuratze-planak aurreikusiz biologia-muga seguruetatik at zeuden espezieentzat, eta urte anitzeko kudeaketa planak ezarriz beste espezieentzat ere, beti ere jasangarritasuna bermatzeko xedearekin. Horrek funtsezko jasangarritasun-paradigma eraldatu zuen: epe motzeko helburuak lortzeko erabaki taktikoetatik, urte anitzeko kudeaketa planen epe luzeko helburuak lortzeko erabaki estrategikoetara.

CFPren azken erreforma, 2013koa, 1380/2013 (EB) Erregulazioaren bitartez gauzatu zen. Politikak oinarritzeko, arrantza-jarduerari buruzko zientzia-ezagutza sendotzeko beharra azpimarratu zen datuen bilketa areagotuz eta espezieen eta flota-ontzien inguruko informazioa partekatzera bultzatuz (Breuer, 2022). Horrez gain, industriaren eta zientzialarien lankidetza sustatu zen eta erabakiak hartzeko eskumena eskualde-mailari utzi zitzaion: Europar Batasuneko legegileek esparru orokorra zehazten dute eta estatu kideek exekuzio-neurriak garatzen dituzte, eskualde-mailako lankidetzan (Breuer, 2022). CFPren azken berrikuspenarekin, kudeaketa helburua MSY adierazlean zehaztu zen (Breuer, 2022): hau da, ahalik eta arrain-bolumenik handiena harrapatzea populazioak ekoizpen maila maximoan mantenduz (Maunder, 2008). Horrez gain, CFPren aurreko berrikuspenean proposatu zen urte anitzeko kudeaketa planen erabilera, erremintarik garrantzitsuenetakoa bilakatu zen Europar Batasuneko arrantza-flotak kudeatzeko. Egun indarrean dauden urte anitzeko kudeaketa plan gehienek arrantzatzen diren espezieak beren MSYan ustiatzeko helburua eta aurrekoa lortzeko neurriak ezartzen dituzte. Neurri horien artean daude TACak eta kuotak, arrantza-esfortzuaren murrizketak eta neurri teknikoak. Harrapatutako arrainak portura eramateko obligazioa 2013ko CFPren erreformak ekarri zuen aldaketarik garrantzitsuenetako bat izan zen (Salomon et al., 2014). Harrapatutako arrainak portura eramateko obligazioaren arabera, harrapaketa-muga duten eta/edo legearen araberako erreferentziazko kontserbazio-tamaina minimo baten menpe dauden espezieen banako guztiak lehorreratuak izan behar dira harrapatuak izanez gero. Erregulazio honen xedea da bazterkinak ezabatzea, ustekabeko edo nahi gabeko harrapaketak saihestuz eta murriztuz (Prellezo and Villasante, 2023).

1.1.2 Arrantza-flota mistoen kudeaketa

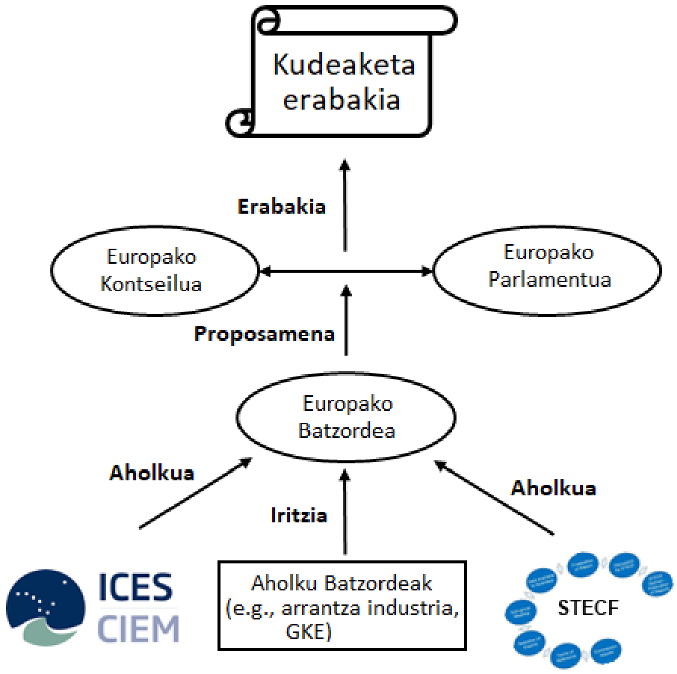
Arrantza-flota mistoetan espezie bat baino gehiago harrapatzen dira aldi berean eta ez dago horien artean bereizteko aukerarik (Wilson and Jacobsen, 2009). Arrantza-flota mistoen harrapaketetan, normalean, produktibitate eta ustiatzearekiko sentikortasun ezberdina duten espezieek egoten dira. Arrantza-flota mistoak kudeatzeak, horrenbestez, erronka gehigarria dakar kudeatzaileentzat, espezieak bereizteko ezintasunaren ondorioz ezinezkoa baita espezie guztien kudeaketa helburuak aldi berean lortzea (García et al., 2019; Matsuda and Abrams, 2006; Ulrich et al., 2016). Litekeena da, espezie zehatz batzuentzat aholkatutako harrapaketak gainditzea beste espezie batzuen arrantza-aukerez baliatzeko, edo espezie batzuei dagozkien arrantza-aukerak erabat ez aprobetxatzea, atalase zorrotzagoak dituzten espezieen arrantza-aukerak ez gainditzeko. Honela, CFPren erronketako bat da urte anitzeko kudeaketa plan eraginkorrak garatzea arrantza-flota mistoentzat (Penas, 2016). Egun, arrantza-flota mistoetarako garatutako urte anitzeko kudeaketa planetan, espezieen arrantza-aukeren aholkua, errendimendu jasangarri maximoan harrapaketek agertzen duten heriotzatasaren inguruan (F_{MSY}) balio minimo eta maximoa emanez zehazten da. F_{MSY} balio-tarte horrek, malgutasun handiagoa eskaintzen du eta, honen xedea harrapatutako espezie guztiak MSY mailara iritsi ahal izatea da (ICES, 2022a). Europan arrantza-flota mistoetarako urte anitzeko lau kudeaketa plan daude egun indarrean. Lehena 2016an onartu zen, Itsaso Baltikoko flotetarako eta espezietarako (EU, 2016). Geroago, 2018an, Ipar Itsasoko flotak eta espezieak kudeatzea xede zuen urte anitzeko kudeaketa plana onartu zen (EU, 2018). 2019an, azkenik, urte anitzeko bi kudeaketa plan sartu ziren indarrean: lehena, Mendebaldeko Uretako eta beren inguruetako uretako flotak eta espezieak arautzeko (EU, 2019b) eta bestea Mendebaldeko Mediterraneo Itsasoko horiek arautzeko (EU, 2019a).

CFPk arrantza-flota mistoen kudeaketari dagokionez landu behar duen beste erronka bat da harrapaketak portura eramateko obligazioak arrantza-ontzien jardueran izan ditzakeen balizko ondorio negatiboak. Izan ere, espezie baten kuota betetzen denean arrantza-ontziak bere jarduera eten beharko du espezie horren kuota ez gainditzeko, arrantzaleek ez badute espezie horren harrapaketak ekiditeko aukerarik beste espezieetako banakoak harrapatzera ateratzen direnean. Eta egoera horretan, ondorioz, beste espezieen kuota bete gabe geratuko litzateke. Arrantza-jarduera muga dezaketen espeziei itoguneko espezie deritze (Schrope, 2010). Arrantza-jardueraren selektibitatean hobekuntzak egitea itomenaren ondorioak gainditzeko irtenbide posible gisa planteatu izan da (Prellezo and Villasante, 2023). Hala ere, arrantza-flota misto baten selektibitate-aldaketek konponbide partziala suposa dezakete harrapaketak portura eramateko obligazioak beren jardueran izan ditzakeen balizko eragina leuntzeko eta harrapaketak portura eramateko obligazioak bideraezina egin dezake arrantza-jarduera (Alzorriz et al., 2016). CFPk, harrapaketak portura eramateko obligazioak arrantza-flota mistoetan izan ditzakeen eragin negatiboei aurre egiteko, malgutasun eta salbuespen ezberdinak proposatu zituen (EU, 2013): de minimis, egoera zehatz batzuetan urteko harrapaketa guztien %5, gehienez, baztertzeko aukera ematen duena; espezieen arteko transferentziak, jomuga diren espezieen kuota %9 haztea ahalbidetzen duena; urteen arteko transferentziak, hurrengo urteko kuotaren %10 harrapatzea ahalbidetzen duena; eta bizirauteko gaitasuna handia duten espezieak baztertzeko aukera ematen duena (EU, 2013). Arrantza-flota mistoetan itoguneko espezieek duten ondorio negatiboa, baina, konpondu gabe dago oraindik.

1.1.3 Europar Batasuneko arrantza-floten kudeaketarako erabakiak hartzeko prozesua

2007an Lisboako Ituna onartu zenetik, CFP exekutatzeko ardura Europako Parlamentuarena eta estatu kideetako ministroez osatutako Europako Kontseiluarena da (Casey et al., 2016). TACak zehaztea, ordea, Europako Kontseiluaren ardura da soil-soilik. Europako Batzordeak CFPri loturiko legedia proposatu eta bultzatzen du. Halaber, bere exekuzioa ere gainbegiratzen du (1.4.Irudia).

CFPren 1380/2013 (EB) erregulazioaren 6.2 artikuluaren arabera, CFPk eskuragarri dauden aholku zientifiko, tekniko eta ekonomikoak hartu behar ditu aintzat (EU, 2013). Europako Batzordeak, horrenbestez, arrantza-flotak kudeatzeari loturiko zientzia-ezagutzarik onenak biltzen ditu hainbat zientzia-erakundetatik (1.4.Irudia). STEFC (Scientific, Technical and Economic Committee for Fisheries) Europako Batzordearen zientzia-batzorde aholkularia da eta ICES (International Council for the Exploration of the Sea) itsas-zientzien gobernu arteko erakunde independente bat da. Azken horren xedea da itsasoko ekosistemen inguruko zientzia-ezagutzak eta -zerbitzuak sustatzea eta partekatzea, eta ezagutza hori, itsasoen jasangarritasuna bermatzeko erabiltzea. Europako Batzordearen azken proposamenak, Europako Kontseiluari eta Europako Parlamentuari bidaltzen zaienak, arrantza sektorean inplikaturiko beste eragile batzuen gomendioak, aholkuak eta/edo iritziak ere hartzen ditu aintzat: arrantza-industriarenak, kontsumitzaileenak edota gobernuz kanpoko erakundeenak (GKE) besteren artean (1.4.Irudia). Inplikaturiko eragile horiek Aholku Batzordetan biltzen dira. Egun, hamaika Aholku Batzorde daude Europan, eskualdearen edo lantzen duten gaiaren araberakoak (adibidez, merkatua, akuikultura, Hego Mendebaldeko urak, pelagikoak, etab).



Irudia 1.4: Arrantza-floten kudeaketari buruzko erabakiak hartzeko prozesua Europar Batasunean.

1.2 ICESen aholkua arrantza-aukeren inguruan: espezie mailatik arrantza-flota mistoetaraino

ICESek arrantza-aukeren inguruan ematen dituen aholkuak espezie-mailan ematen ditu, espezie mailako kudeaketa helburuak sostengatzeko helburuz. Aholkuek zuhurtasunaren printzipioa eta MSYa lortzeko helburua integratzen dituzte (ICES, 2022a). Arrantza-flota mistoari loturiko kudeaketa helburuen faltan, ICESek ez ditu arrantza-flota misto espezifikoek espezie zehatzak harrapatzeko dituzten aukeren inguruko aholkuak ematen. Baina, arrantza-flota mistoen inguruko kontsiderazioak egiten ditu: espezie-mailan ematen diren harrapaketa aholkuek arrantza-flota mistoen jardueran eta espezieen jasangarritasunean duten eragina aztertuz, flota-dinamikaren inguruko agertoki ezberdinak erabiliz. Horri esker, espezie ezberdinen aholkuak urte zehatz horretan, espezieen egoeran duten eragina ebalutzen dute flota-dinamika ezberdinetarako eta floten jarduera gehien mugatzen duten espezieak identifikatzen dituzte.

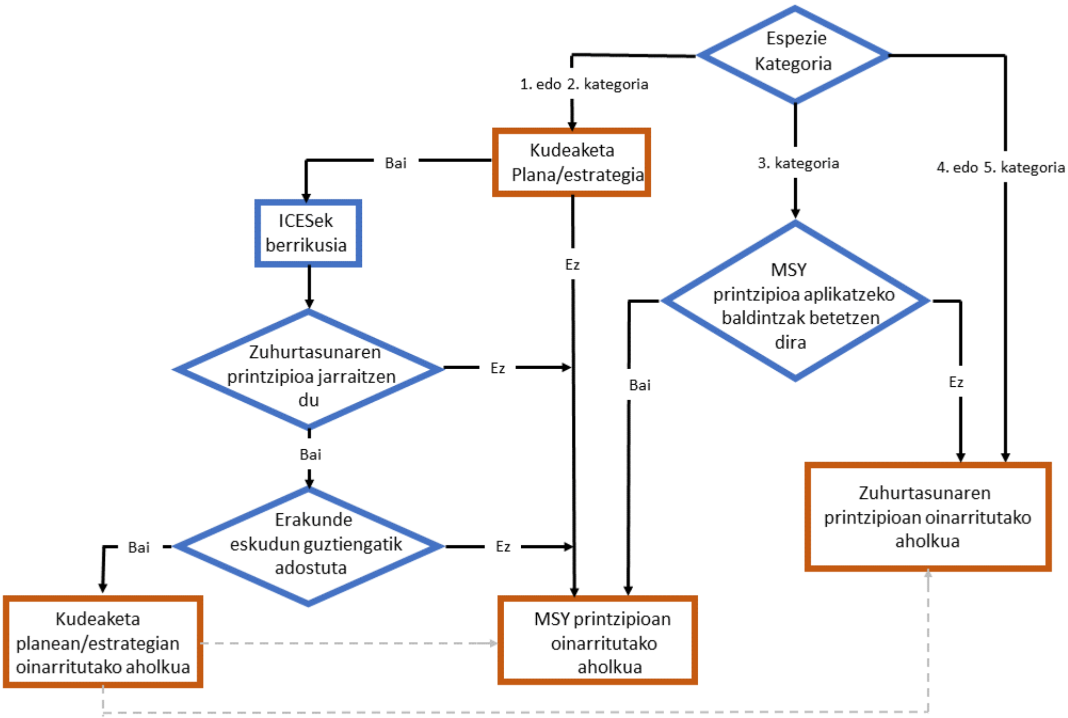
1.2.1 Espezie mailan emandako aholkuak

2012an ICESek, espezie bakoitzaren arrantza-aukerei buruzko aholku kuantitatiboa emateko, esparru bat garatu zuen (ICES, 2012). Esparru horretan, espezieak sei kategoria nagusitan sailkatzen dira datu eskuragarritasunaren eta ezagutzaren arabera. 1.kategoriak ebaluazio kuantitatiboak dituzten espezieak hartzen ditu barnean, baina datuen eskuragarritasuna eta kalitatea baxuagoak dira kategorietan behera egin ahala (1.1.Taula). 3. edo goragoko kategorietan dauden espezieen inguruan eskuragarri dauden datuak mugatuak dira eta datu-mugatuak dituzten espezieak deritze (ICES, 2022a). 2020.urtean ICESek 260 espezie baino gehiagori buruzko banakako aholkuak eskaini zituen, urtebeteko edo bi urteko maiztasunarekin. Espezie horietatik %60 baino gehiago datu-mugatuak zituzten espezieak ziren (ICES, 2020c).

Taula 1.1: Espezie kategoriak (Kat) eskuragarri dagoen ezagutzaren arabera (ICES, 2022a).

Kat	Deskribapena
1	Ebaluazio kuantitatiboak dituzten espezieak; barnean daude ebaluazio analitiko integralak edo adinaren/luzeraren zein produkzio-ereduen arabera egindako aurreikuspenak dituzten espezieak.
2	Kualitatiboki soilik tratatuak izaten diren ebaluazio analitikoak eta aurreikuspenak dituzten espezieak, bai eta soberakin-produkzioko ereduak dituzten espezieak ere; barnean daude zio ezberdinen ondorioz harrapaketen heriotza-tasari, errekrutatzeari eta biomasari loturiko joeran adierazle diren ebaluazio eta aurreikuspen kuantitatiboak dituzten espezieak ere.
3	Egindako ikerketen edo esplorazioen eskutik joerak zehazten dituzten ebaluazioak dituzten espezieak; barnean daude joerak zehazteko ariketen edo beren bizitzari buruzko informazioaren ondorioz heriotza-tasari, errekrutatzeari eta biomasari buruzko joera fidagarriak markatzen dituzten ebaluazioak dituzten espezieak ere.
4	Zigala espezieak, balizko abundantziaren informazioa iradokitzeko aukera ematen dutenean; eta denboran zehar egondako harrapaketei jarraiki, MSY gutxi gorabehera kalkulatzeko aukera ematen duten espezieak. Arrazoizko zientzia-zioak daude funtzio-unitateek ematen duten bizi-historiari eta dentsitateari buruzko informazioa aholkuak emateko erabiltzeko.
5	Lehorreratzeen edo epealdi labur batean egondako harrapaketen datuak agertzen dituzten espezieak.
6	Lehorreratze hutsalak dituzten edo kopuru txikian harrapatu diren espezieak, nahi gabean harrapatu diren eta baztertuak izango diren espezieak; barnean daude lehorreratzeak baztertzeak baino askoz txikiagoak dituzten espezieak, bai eta jomuga diren beste espezie batzuk harrapatzerakoan nahigabean harrapatzen diren espezieak ere.

ICESek kudeaketa aholkua eskaintzeko, metodo eta harrapaketak ezartzeko arau (Harvest Control Rule, HCR) ezberdinak aplikatzen ditu kategoria bakoitzeko espezieentzat. Oro har, HCR arauak, xede zehatz bat lortzeko arrantza-baliabideen harrapaketa-mugen inguruko aholkua emateko erabiltzen diren matematika-formulak dira. ICESek, 1. eta 2.kategorietan dauden espezieen harrapaketa aholkua, zuhurtasunaren printzipioari jarraitzen dioten kudeaketa planak edo estrategiak aintzat hartuz ematen du (1.5.Irudia). Erakunde eskudunak ez baditu plan horiek baimendu edo ICESen ustez planek ez badiote zuhurtasunaren printzipioari jarraitzen, 1. eta 2.kategorietan dauden espezieen aholkua MSY ikuspegia aintzat hartzen duen HCR araua erabiliz eraikiko da (ICES, 2022a). 3.kategoriako espezieen arrantza aholkuak, adierazle enpirikoetan (abundantzia-indizeetan, adibidez) oinarritzen diren HCRetatik kalkulatzen dira. HCR horien harrapaketa aholkuak, espeziearen abundantzia-indizean ikusitako joeran oinarritzen dira. 2022.urtean datu gutxi behar dituzten HCR alternatiboak garatzen hasi zen ICES (ICES, 2022a), adierazle enpirikoetan eta MSY proxyetan oinarritutako aholkuak eskaini ahal izateko 3.kategorian dauden espezieentzat (1.5.Irudia). 4. edo goragoko kategoriatan dauden espezieentzat, zuhurtasunaren printzipioa oinarri hartuta ematen dira aholkuak (1.5.Irudia). Printzipio horren arabera, arrantza-aukerei buruzko aholkua, aldian behin harrapaketa aholkua %20an murriztuz ematea proposatzen da, zuhurtasunaren printzipioa aplikatuz eta inolako erreferentzia-punturik erabili gabe. Zenbat eta mugatuagoa izan espezieen egoerari buruzko informazioa, are eta kontserbadoreagoa izan beharko luke emandako aholkuak eta zuhurtasun-marjina handiagoa aplikatu beharko litzateke (ICES, 2022a).



Irudia 1.5: ICES aholkuen oinarria azaltzen duen fluxu-diagrama. Marra gris etenek adierazten dute kudeaketa planen inguruko aholkuek ICES MSY ikuspuntuari edo zuhurtasunaren printzipioari jarraitzen diotela ICES (2022a).

1.2.2 Arrantza-flota mistoen inguruko kontsiderazioak

Arrantza-flota mistoen inguruko kontsiderazioetan, espezie-mailan ematen diren harrapaketa aholkuek arrantza-flota mistoen jardueran eta espezieen jasangarritasunean duten eragina ebaluatzen da flota-dinamikei buruzko hipotesi ezberdinak aplikatuz. Honek, flota-dinamikako agertoki ezberdinak aztertzean datza elkarrekintza teknikoen (harrapatutako espezieen artean) egungo ezagutza eta espezie-mailan ematen diren harrapaketa aholkuak aintzat hartuz. Kontsiderazio horien helburua ez da arrantza aholku bakar eta optimo bat ematea arrantza-flota mistoetarako, baizik eta espezie mailan ematen diren harrapaketa aholkuek, arrantza-flota mistoen kontestuan, sor ditzaketen desoreken berri ematea. Desoreka horiek gertatzen dira, espezie guztien harrapaketa aholkuak ezin direlako aldi berean bete. 2012.urtean, eta lehen aldiz Europan, ICESek arrantza-flota mistoen inguruko kontsiderazioak eskaini zituen Ipar Itsasoko arrantza-flota demertsalerako (ICES, 2021j). Ondoren, 2015ean, 2016an eta 2020an arrantza-flota mistoen inguruko kontsiderazioak garatuko zituen Zeltiar Itsasorako, Iberiar penintsulako uretarako eta Bizkaiko Golkorako, hurrenez hurren (ICES, 2021j). 2022an arrantza-flota mistoen inguruko kontsiderazioak Irlandako Itsasorako aurkeztu ziren lehen aldiz (ICES, 2022e). Horrez gain, arrantza-flota mistoen inguruko kontsiderazioak Itsaso Baltikorako garatzen ari dira egun (ICES, 2022e,f).

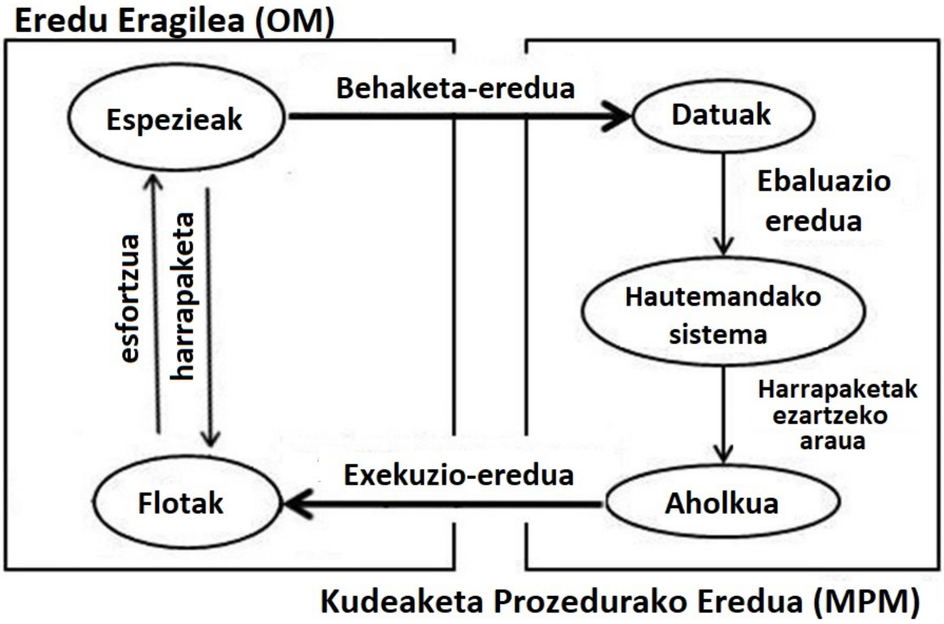
1.3 Kudeaketa estrategiaren ebaluazioa

Espezie bakarreko arrantza-flotak kudeatzeko aholku zientifikoa emateko ikuspegi tradizionala, egiazko sistema-dinamiken inguruko hipotesi bakarrean oinarritzen da. Hipotesi hori normalean, datuetara hoberen egokitzen den ebaluazio-eredutik dator, bai eta populazioaren eta arrantza-flotaren inguruko ezagutzatik ere (Punt and Donovan, 2007). XX.mende erdialdean, zenbait nazioarteko arrantza-flota komertzial kolapsatu ziren, eta kolokan jarri zuten ebaluazio- eta kudeaketa-eredu matematikoren erabileraren eraginkortasuna arrantza-aukeren inguruko aholkuen oinarri gisa (Schnute and Richards, 2001). Zientzia-komunitateak ezin izan du flota horien kolapsoaren arrazoi zehatza zehaztu eta adostu, baina aditu gehienek aitortu dutenez, arazo nagusietako bat zen, ez zela ziurgabetasuna kontuan hartu kudeaketa prozesuan (Punt et al., 2016; Schnute and Richards, 2001).

Ikuspegi tradizionalaren alternatiba gisa, XX.mende amaieran lehen aldiz, kudeaketa estrategiaren ebaluazioa (Management Strategy Evaluation, MSE) erabili zen. MSEak kudeaketa aholkuen inguruko erabakiak hartzeko prozesuan ziurgabetasuna sartzea ahalbidetzen du. MSEak kudeaketa estrategia ezberdinek espezieen eta floten jasangarritasunean duten eragina aztertzea ahalbidetzen du, sistema-dinamiken inguruko hipotesi ezberdinak aplikatuz, ziurgabetasun-iturri ezberdinak barne. MSEren xedea da, aurretik zehaztutako kudeaketa helburuak lortzean dituen eta sistemaren ziurgabetasunetara hobekien egokitzen den kudeaketa estrategia identifikatzea (Punt et al., 2016). MSEaren erabilerak paradigma-aldaketa suposatu zuen. Eskuragarri zeuden datuetara hoberen egokitzen zen eredua bilatu ordez, xedea arrantza-sistemaren ziurgabetasunera hoberen egokitzen den kudeaketa prozedura bat aurkitzea izatera iritsi zen.

MSEaren aintzindaria IWC (International Whaling Commission) izan zen 1980an eta Smithek sartu zuen arrantza-floten hiztegian 1994an (Smith, 1994). Arrantza-floten ikerketetan, baina, MSEren erabilera XXI.mendean zabaldu zen, gaitasun konputazionalean izandako iraultzari esker. Egun, mundu osoan erabiltzen da, urte anitzeko kudeaketa plan bat ebaluatzeko modurik egokientzat jotzen baita (Punt et al., 2016). MSEk, horrez gain, zientzia-aholkuen gardentasuna eta zientzialarien eta inplikatutako eragileen arteko komunikazioa hobetzen ditu (deReynier et al., 2010), erabakiak hartzeko prozesuan politikak duen eragina murriztuz (Punt and Donovan, 2007).

MSErako eskuragarri dauden software ezberdinen artean, FLBEIA (Bio-Economic Impact Assessment using FLR libraries) (García et al., 2017) Europako arrantza-flota mistoak kudeatzeko eskuragarri dagoen simulazio-eredu bio-ekonomikoetako bat da (ICES, 2022f). FLBEIA eredua R-n dago garatua (R Core Team, 2022), FLR liburutegiak erabiliz (Kell et al., 2007). FLBEIA eredua elkarri eragiten dioten bi osagai nagusiz osatua dago (1.6.Irudia): eredu eragilea (Operating Model, OM) eta kudeaketa prozedurako eredua (Management Procedure Model, MPM). OMk benetako sistema simulatzen du eta espezieak, flotak eta beren arteko elkarrekintzak hartzen ditu barnean. MPM kudeaketa prozesua simulatzen duen ereduaren zatia da eta ondokoak biltzen ditu: datuak, hautemandako sistema eta aholkua, behaketa-ereduak, ebaluazio-ereduak eta HCRk sortua, hurrenez hurren. OM eta MPMren arteko lotura behaketa-ereduaren bitartez gauzatzen da. Behaketa-ereduak OMtik hautemandako datuak sortzen ditu, MPM ebaluazio-ereduak erabil ditzan. MPM eta OMren arteko lotura exekuzio-ereduaren bitartez gauzatzen da. Exekuzio-ereduak, MPMk emandako kudeaketa aholkua errealitatean nola aplikatuko den deskribatzen du. Eredua arrantza-flota kudeatzeko sisteman dauden ziurgabetasun iturri nagusiak kontuan hartzeko diseinatuta dago, Francis and Shotton (2011)k deskribatu bezala.



Irudia 1.6: Kudeaketa estrategia ebaluatzeko esparruaren kontzeptu-diagrama. Garcia et al. (2013)en lanetik egokituta.

Azken aldian, eztabaida dago simulazio-eredu baten aplikazioa noiz defini daitekeen MSE gisa eta noiz ez. Autore batzuen ustez, kudeaketa estrategien simulazio-ereduak, HCRtik eratorritako harrapaketa aholkuak errealitatean akatsik gabe ezartzen direnean, ezin dira MSEtzat hartu (Punt et al., 2016). Simulazio-eredua espezie bakar beteri aplikatzen zaionean, aldea, hautemandako sistema lortzeko kudeaketa prozeduran, ebaluazio-eredu bat sartzearekin lotu ohi da.

MSE eredua espezie bakarrei aplikatu zitzaien hasiera batean (e.g., Garcia et al., 2011; Polacheck et al., 1999). Azken hamarkadetan, ordea, ekosisteman oinarritutako arrantza-floten kudeaketaranzko (Ecosystem-Based Fisheries Management, EBFM) norabidean, arrantza-floten kudeaketaren ikuspegi holistiko baten beharraren eskutik, espezie anitzeko eta flota anitzeko MSE aplikazioa bultzatu da (e.g., García et al., 2019; Simons et al., 2014). Horrez gain, MSE ereduaren OMa muturreko ekosistema-eredu integral batean oinarritzeko saiakerak ere egin dira (e.g., Dichmont et al., 2016b; Fulton et al., 2016), EBFM trantsizio gisara (Lidström and Johnson, 2020).

1.3.1 Arrantza-flota mistoen MSE eraikitzeko erronkak

Arrantza-flota mistoek, espezie ugari harrapatzen dituzte eta ezinezkoa da espezie guztiak esplizituki simulazio-ereduan sartzea horrek suposatzen duen lan karga handiagatik, behar diren baliabide konputazionalengatik eta eskuragarri dauden datu mugatuengatik. Hala ere, simulazioan sartu behar diren espezieek, adierazgarritasun nahikoa bermatu behar dute flotaren jarduera deskribatzeko. Horrek bermatzen du kudeaketa estrategia batek jasankortasun biologikoan duen eragina zehatz ebaluatu ahal izatea (Ulrich et al., 2011). Eredu eragilean sartutako espezieek zehaztuko dute arrantza-jarduera zeinen ongi deskribatu daitekeen eta zer ebaluatu daitekeen jasankortasun biologikoari dagokionez. Ondorioz, espezie anitzeko MSE bat eraikitzeko erronketako bat da simulazioan sartu beharreko espezieak identifikatzea. Simulazio-ereduan sartu beharreko espezieak aukeratu ondoren, hurrengo urratsa espezieen populazio-dinamika zehaztea da, MSEn sartu ahal izateko. Arrantza-flota mistoetan harrapatutako espezie gehienak nahigabean harrapatzen dira eta baztertuak izaten dira. Espezie hauentzat eskuragarri dauden datuak oso mugatuak dira. Espezie anitzeko MSE ikerketa gehienek datu-ugariko espezieetan jarri dute arreta eta barne hartu dituzten datu-mugatuak dituzten espezie bakanak, era sinplistegian tratatu dituzte (e.g., Prellezo et al., 2016; STECF, 2015). Datu-mugatuak dituzten espezieen dinamika MSEn sartzea, simulazio-ereduak eraikitzeko erronka handienetako bat da.

1.3.1.1 Espezieak aukeratzea

Simulazioan sartu beharreko espezieen aukeraketa, jasangarritasunaren hiru habeen testuinguruan egin beharko litzateke: gizarte-ekitatea, bideragarritasun ekonomikoa eta babes ekologikoa (WCED, 1987). Baina, askotan, simulazio-ereduan sarturiko espezieak, aurreko hiru alderdietako bakar bat oinarri hartuta aukeratzen dira. arrantza-ontzientzat ekonomikoki garrantzitsuenak diren espezieak aintzat hartzen dituzten MSE ikerketak daude (e.g., Hamon et al., 2007; Prellezo et al., 2016) bai eta ekologikoki babestuak izan behar duten espezietan zentratzen direnak ere (e.g., Bastardie et al., 2010; Punt et al., 2001).

Hobday et al. (2011, 2007)k arrantzaren eraginez sortzen den arrisku ekologikoa ebaluatzeko metodologia bat garatu zuen. Metodologia honek, arrisku ekologikoa ebaluatzeko ereduak hiru maila desberdinetan banatzen ditu ereduak behar dituen datuen arabera. 3.mailan erabat kuantitatiboak diren ereduak daude; 2.mailan, eredu erdikuantitatiboak direnak eta 1.mailan arriskuaren analisi kualitatiboa egitea ahalbidetzen duten ereduak daude (Hobday et al., 2011, 2007). Produktibitate-suszeptibilitate analisia (Productivity-Susceptibility Analysis, PSA) datu-eskari baxuko eredu erdikuantitatiboa da eta arrisku ekologikoa ebaluatzeko esparruan 2. mailan kokatzen da (Hobday et al., 2011, 2007). PSAk, arrisku potentziala zehazteko bi ezaugarri hartzen ditu aintzat: produktibitatea (murriztearen ostean aurreko egoerara bueltatzeko espezieak duen gaitasuna) eta suszeptibilitatea (arrantza-jarduerek espeziearen jasangarritasunean duten eragin potentziala). PSA ereduaren oinarria da, produktibitate baxuko eta suszeptibilitate altuko egoeretan arrisku potentzial handia dagoela espeziearentzat (Hobday et al., 2007). PSAk bere balioa frogatu du espezie ezberdinen arrisku potentzial erlatiboa esparru berean kalkulatzeko, espezieen artean datu eskuragarritasunari dagokionean egon daitezkeen aldeen gainetik (Cortes et al., 2015; Hobday et al., 2011; Lucena-Fredou et al., 2017; Ormseth and Spencer, 2011).

Osio et al. (2015)k Mediterraneoko arrantza-baliabide demertsalak lehenesteko metodo bat proposatu zuen, datuak bildu eta ikertzeko behar handiena zuten espezieak identifikatzeko. Espezieak lehenesteko, ondoko alderdiak izan zituen kontuan: lehorreratzeei egiten zieten ekarpena, beren batez besteko prezioa kiloko; eta beren arrisku ekologikoa arrantzaren eraginarekiko PSA erabiliz.

1.3.1.2 Datu-mugatuak dituzten espezieen populazio-dinamika zehaztea

Espezie anitzeko MSE aplikazioetan, espezie enblematikoenak esplizituki sartu behar dira, datu-mugatuak dituzten espezieak barne. Datu-mugatuak dituzten espezieak sartu dituzten zenbait MSE ikerketetan, bizi-historiaren parametroak, lehen urte historikoan zegoen biomasa-mailaren inguruko hipotesia eta ustiapen historikoaren inguruko hipotesia erabili zituzte populazio-dinamikaren ekuazioak parametrizatzeko (e.g., Carruthers et al., 2014; Fischer et al., 2020). Beste aukera bat da datu gutxi behar dituen ebaluazio-eredu bat erabiltzea datu-mugatuak dituzten espezieak simulazio eredu batean sartzeko. Esaterako, biomasan oinarritutako ereduak (e.g., Harlyan et al., 2019; Mildenberger et al., 2022), edo adinean oinarritutako ereduak (e.g., Geromont and Butterworth, 2014; Huynh et al., 2020b).

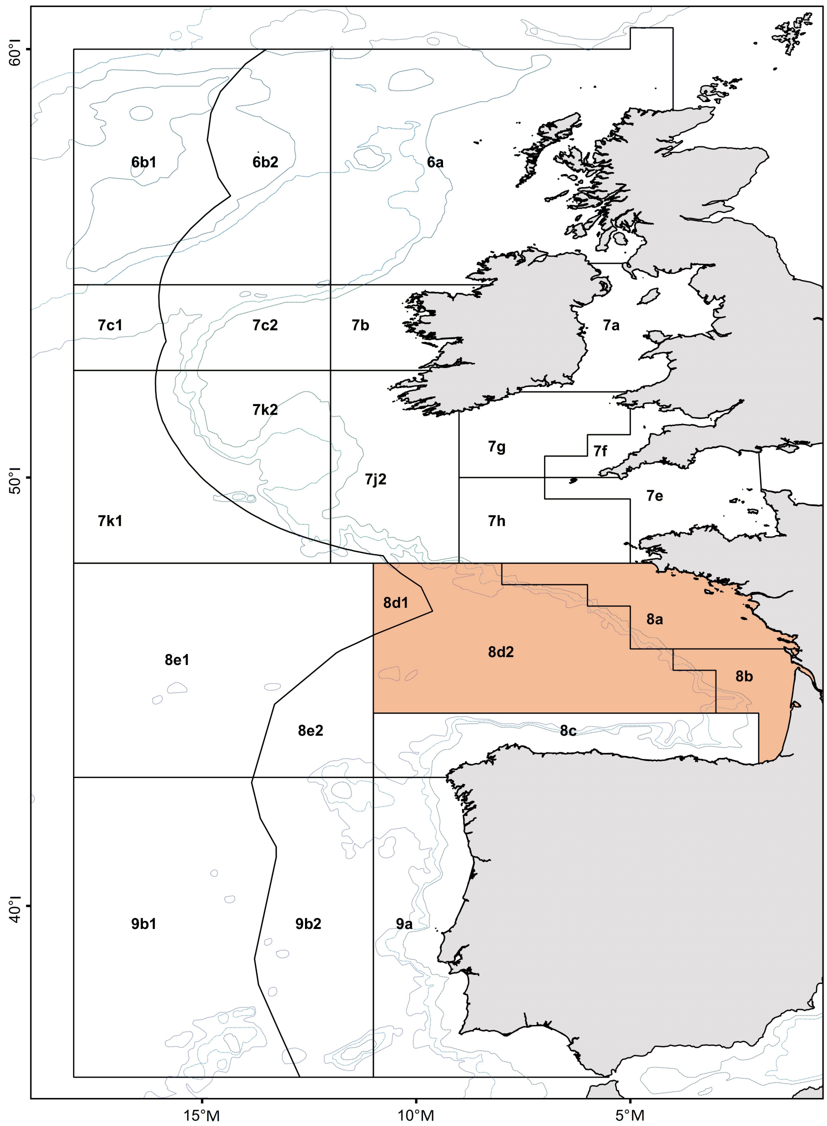
Azken urteotan biomasan oinarritutako ereduen populartasunak gora egin du ICES komunitatean datu-mugatuak dituzten espezieen egoera aztertzeko (ICES, 2020c). ICESek datu-mugatuak dituzten espezieen populazio-dinamikak ebaluatzeko gehien erabiltzen dituen biomasa-ereduak ondokoak dira (ICES, 2020c): SPICT (Surplus Production model In Continuous Time, Pedersen and Berg, 2017, -ek garatua) eta JABBA (Bayesian State–Space biomass-dynamics model, Winker et al., 2018, -ek garatua). Biomasan-ereduek espezieen dinamika era sinplean irudikatzen dute. Adinaren konplexutasunari ez ikusiarena egin eta abundantzian egon daitezkeen aldaketak ondoko alderdietan oinarrituz azaltzen saiatzen dira: biomasa-galera arrantzaren ondorioz, biomasa aurreko urtean eta biomasan egondako hazkundea. Halaber, hipotesi sendoak egiten dituzte modelatutako sistemaren izaeraren eta erabilitako datuen inguruan (ikus Punt and Hilborn, 1996, -en lana hipotesi horiek ezagutzeko). Emaitzak, ondoko bi aldagaiekiko sentikorrak dira: adin-oinarri iragankorren eraginekiko (Punt and Szuwalski, 2012) eta produkzio-funtzioarekiko (Maunder, 2003). Oro har, espezieak ebaluatzeko adinean oinarritutako ereduak erabiltzea biomasan oinarritutako ereduak erabiltzea baino hobe da, populazioaren- eta arrantza-dinamikei buruzko datu gehiago sartzeko aukera ematen baitute: adinaren araberako selektibitatea, errekrutatzea eta banakoen hazkundea esaterako (Maunder, 2003). Espezieak ebaluatzeko eredu garaikide gehienak adinean oinarritutako ereduak dira (Punt et al., 2013). Baina ez da erraza eredu horiek datu-mugatuak dituzten espezieei aplikatzea, gehienek adinaren araberako harrapaketen datuak behar baitituzte eta datu horiek ez baitdaude eskuragarri datu-mugatuak dituzten espezie gehienentzat.

Espeziearen murrizteari buruzko analisia (Stock Reduction Analysis, SRA) adinaren araberako harrapaketen daturik ez dagoen kasuetarako sortu zen (Huynh et al., 2020a; Kimura and Tagart, 1982; Walters et al., 2006). SRA datu gutxi behar dituen adinean oinarritutako ebaluazio-eredu bat da. SRA ereduak, guztizko harrapaketen eta abundantzia-indizearen inguruan eskuragarri dauden datuak eta bizi-historiari buruzko datuak konbinatzen ditu populazio-dinamikaren eta ustiatze-mailaren inguruko kalkulu historikoak lortzeko. SRA ereduak abundantzia-indizearen selektibitatea, arrantza-jardueraren selektibitatea, biologia-parametroak eta errekrutatze-desbideratzeen desbideratze estandarra jakinda, behatutako datuak sortuko lituzketen hasierako biomasaren, harrapaketen heriotza-tasaren eta errekrutatze-mailaren konbinazio posibleak kalkulatzen ditu (Huynh et al., 2020a). SRA eredua aplikatzeko behar diren gutxieneko datuak dira urteko harrapaketa guztiak eta guztizko biomasaren abundantzia-indizea. Datu horiez gain, beste datu batzuk ere sar daitezke SRA ereduan: harrapaketen batez besteko luzera, luzeraren araberako harrapaketak, adinaren araberako harrapaketak eta adinaren araberako indizea. Halaber, parametro biologiko eta datu-iturri guztietan ziurgabetasuna sartzea ahalbidetzen du. SRA eredua, datu-mugatuak dituzten espezieen populazio-dinamikak simulazio-ereduetan sartzeko erabili izan da (Hordyk et al., 2019; Huynh et al., 2020b). Hordyk et al.

(2019)ek SRA ereduaren errendimendua aztertu zuen biologia-parametroetan alborapena eta behaketa- eta eredu-hipotesietan akatsak sartuz. Hala ere, SRA ereduak populazio-dinamiken kalkulu zehatzak lortzeko duen gaitasuna datuen eskuragarritasun egoera ezberdinetan ez da inoiz kuantitatiboki ebaluatu.

1.4 Ikerketarako zioak

Bizkaiko Golkoa, ondo bereizita dagoen unitate geomorfologiko bat da. Bizkaiko Golkoa, Ipar-mendebalderantz orientatua dago eta Ipar-ekialdeko Itsas Atlantiko epelean dago kokatuta, Zeltiar Itsasotik hegoaldera (Lavín et al., 2006) (1.7.Irudia). Bizkaiko Golkoko mugak dira: Espainiako itsasertza hegoaldean eta Frantziako itsasertza ekialdean. Ur boreal hotzen eta probintzia biogeografiko epeleko ur beroen arteko trantsizio-eskualdea osatzen du (Lavín et al., 2006) eta, ondorioz, bere inguruko urak baino biodibertsitate handiagoa dauka, honek Bizkaiko Golkoa, arrantza-flota mistoen jarduera gauzatzeko ingurune ezin hobea bilakatzen du (Prellezo et al., 2016).



Irudia 1.7: Europa Mendebaldeko itsas eremuak, ICESen banaketaren arabera. Itzal laranja duen eremua ikertutako Bizkaiko Golkoko eremua da.

Bizkaiko Golkoan harrapatutako espezieen kudeaketa CFPren eta eskualdeko legediaren arabera egiten da. Eremu honetan jarduten duten arrantza-flotak kudeatzeko erreminta nagusiak TACak eta kuotak dira (ICES, 2021a). Baina, badaude zenbait neurri tekniko ere, 2019/1241 (EB) Erregulazioak ezarrita, espezie eta aparailu batzuentzat, arrantza-baliabideak kontserbatu eta itsasoko ekosistemak babesteko. Neurri tekniko horien artean daude lehorreratzen diren banakoen tamaina minimoak, sare-begien tamaina mugak eta babestutako itsas eremuak. Kontserbazioa bermatzeko erreferentziazko tamaina minimoa 41 espezierako ezarrita dago Bizkaiko Golkoan. Sare-begiaren tamaina mugatua dute arraste ontziek, sare estatikoek eta noraezeko sareek, aparailuaren jomuga diren espezieen arabera tamaina ezberdinetan. Harrapaketak portura eramateko obligazioa 2015ean ezarri zen espezie pelagikoentzat, espezie demertsalentzat aldiz 2016an, eta 2019ko urtarriletik aurrera era integralean dago erregulazioa indarrean Bizkaiko Golkoan (ICES, 2021a). Salbuespen gisa, aipatu obligazio horretatik kanpo geratu direla arraiak, baztertuak izan ondoren bizirauteko gaitasun handia dutelako. Horrez gain, legatza, oilarra, zapoak, mihi-arraina, berdela eta txitxarro beltza, de minimis salbuespenaren menpe daude, honela, urtean harrapatutako espezie hauen bolumen guztiaren %5a baztertua izan daiteke (EU, 2023). Bizkaiko Golkoko espezie demertsal batzuk Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planean sartuta daude (EU, 2019b). Planaren xedea da, nahigabeko harrapaketak murriztea, arrantzak itsasoko ekosisteman dituen eragin negatiboak minimizatzea, eta CFPak ezarritako helburuak lortzen laguntzea zuhurtasunaren printzipioa arrantza-floten kudeaketan aplikatuz eta harrapatutako populazioak MSY mailatik gora daudela bermatuz.

Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoek tradizio luzea dute (Prellezo, 2010; Prellezo et al., 2016) eta Frantziako eta Espainiako ontziek osatzen dute gehienbat (eremuko harrapaketa guztien %86aren erantzule dira): Bizkaiko Golkoan ematen diren harrapaketa guztien %53 Frantziako 1.500 ontzi ingururi dagokio eta beste %33a eremu horretan jarduten duten Espainiako 57 ontziri. Beste harrapaketak ondoko herrialdeetako ontziei dagozkie: Belgika, Alemania, Danimarka, Irlanda, Herbehereak, Polonia, Portugal eta Erresuma Batua. Frantziako eta Espainiako ontziak hondoko eta erdi-mailako arraste ontziak, inguratze-arrantza ontziak eta tretzaontziak dira gehienbat. Ontzi horiek harrapatzen dituzten espezie nagusiak ondokoak dira: legatza, zapoa, oilarra, mihi-arraina, zigala, lupia arrunta eta abadira (ICES, 2021a). Baina ondokoak ere arrantzatzen dituzte: txitxarro beltza, berdela, bakalada, lupia arrunta, txibia eta txipiroiak. Harrapatutako espezieak arrantza-eremuaren, arrantza-sakoneraren eta erabilitako aparailuaren menpe daude. Oro har, Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoek 150 espezie baino gehiago harrapatzen dituzte (Altuna-Etxabe et al., 2020; Briton et al., 2020). Baina espezie horietako laurdenak baino ez daude TAC eta kuota-sisteman sartuta. Hauen artean legatza, oilarra, zapo zuria, zapo beltza, liba, zigala, mihi-arraina, berdela, txitxarro beltza, bakalada, abadira, arraia zerra, arraia gastaka eta mosaiko arraia TAC eta kuota-sisteman sartuta daude (EU, 2022) (1.2.Taula). Lehen hamaika espezieak ICES Bizkaiko Golkoko arrantza-flota mistoen inguruko kontsiderazioetan kontutan hartzen dira (ICES, 2022b). Lehen zazpi espezieak gehi lupia arrunta, bestetik, Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planean sartuta daude (EU, 2019b). Harrapatutako beste espezie gehienak datu-mugatuak dituzten espezieak dira, beren egoera ezezaguna da, eta inolako kuota-mugarik gabe arrantzatzen dira (1.2.Taula).

Taula 1.2: Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoek harrapatzen dituzten eta TAC eta kuota-sisteman eta/edo Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planean sartuta dauden espezieen inguruko informazioa. Espeziearen izen arrunta eta zientifikoa, banaketa-eremua ICES nomenklatura erabiliz, ICES datu-kategoria (Kat), aholku mota (MSY ikuspegia edo zuhurtasunaren printzipioa (PA)), MSY erreferentzia-puntuekiko egoera, eta espezieak TAC eta kuota-sisteman (TAC) eta/edo Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planean (MP) eta/edo ICES Bizkaiko Golkoko arrantza-flota mistoen inguruko kontsiderazietan (MF) dauden ala ez adierazten duen informazioa eta erreferentzia. Egoera zutabean, grisak erreferentzia-puntuak ezezagunak direla adierazten du, berdeak espeziea F_{MSY}en azpitik harrapatzen dela eta/edo $B_{trigger}$tik gorako tamaina duela adierazten du eta, gorriak, espeziea F_{MSY}tik gora harrapatzen dela eta/edo $B_{trigger}$tik beherako tamaina duela.

Izen arrunta	Izen zientifikoa	Banaketa-eremua	Kat	Aholkua	Egoera		Kudeaketa			Erreferentzia
					F	B	TAC	MP	MF	
Legatza	*Merluccius merluccius*	4, 6 eta 7 azpiguneetan, eta 3a eta 8abd dibisioetan	1	MSY	✓	✓	X	X	X	ICES (2021c)
Oilarra	*Lepidorhombus whiffiagonis*	7b-k eta 8abd dibisioetan	1	MSY	✓	✓	X	X	X	ICES (2021g)
Zapo zuria	*Lophius piscatorius*	7 azpigunean, eta 8abd dibisioetan	1	MSY	✓	✓	X	X	X	ICES (2021o)
Zapo beltza	*Lophius budegassa*	7 azpigunean, eta 8abd dibisioetan	1	MSY	✓	✓	X	X	X	ICES (2022c)
Liba	*Merlangius merlangus*	8 azpigunean, eta 9a dibisioan	5	PA	?	?	X	X	X	ICES (2021p)
Zigala	*Nephrops norvegicus*	8ab dibisioetan	1	MSY	✓	?	X	X	X	ICES (2021h)
Mihi-arraina	*Solea solea*	8ab dibisioetan	1	MSY	✗	✗	X	X	X	ICES (2021m)
Berdela	*Scomber scombrus*	1-7 eta 14 azpiguneetan, eta 8a-e eta 9a dibisioetan	1	MSY	✓	✓	X		X	ICES (2021f)
Txitxarro beltza	*Trachurus trachurus*	8 azpigunean, eta 2a, 4a, 5b, 6a eta 7a-c,e-k dibisioetan	1	MSY	✓	✗	X		X	ICES (2021d)
Bakalada	*Micromesistius poutassou*	1-9, 12 and 14 azpiguneetan	1	MSY	✗	✓	X		X	ICES (2021b)
Abadira	*Pollachius pollachius*	8 azpigunean, eta 9a dibisioan	5	PA	?	?	X		X	ICES (2021i)
Arraia zerra	*Leucoraja naevus*	6-7 azpiguneetan, eta 8abd dibisioetan	3	PA	?	?	X			ICES (2020a)
Arraia gastaka	*Raja clavata*	8 azpigunean	3	PA	?	?	X			ICES (2020d)
Mosaiko arraia	*Raja undulata*	8ab dibisioetan	6	PA	?	?	X			ICES (2020e)
Lupia arrunta	*Dicentrarchus labrax*	8ab dibisioetan	1	MSY	✓	✓		X		ICES (2021k)

2015ean, STEFCk Mendebaldeko Urak eta beren ingurukoak kudeatzeko urte anitzeko planak Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen aktibitatean eta beren arrantza-baliabideen jasankortasunean izan dezakeen eragina ebaluatu zuen (STECF, 2015). Bi simulazio-eredu exekutatu zituzten: bata Frantziako flotak kontuan hartuz, Espainiako flotei lotuta bestea. Frantziako simulazio-ereduan, eredu eragileak 17 espezie hartzen zituen barnean, Frantziako flotaren lehorreratze guztien %56a eta diru-sarrera guztien %68a suposatzen zutenak (STECF, 2015). Espainiako simulazio-ereduan erabilitako eredu eragilean 12 espezie sartu ziren, Espainiako flotaren harrapaketa guztien %81 eta diru-sarrera guztien %88 suposatzen zutenak (STECF, 2015). Espezie horietatik, Frantziako ereduan hiru espezie (legatza, mihiarraina eta zigalak) eta Espainiako ereduan bi espezie (legatza eta oilarra) baino ez ziren simulatu adinean oinarritutako eredu bat erabiliz (STECF, 2015). Beste espezie guztiak simulatzeko esfortzu unitateko harrapaketa (Catch Per Unit Effort, CPUE) konstantearen ikuspegia erabili zen. Horren arabera, harrapaketek lotura lineala dute esfortzuarekin eta biomasarekiko independenteak dira (STECF, 2015). CPUE konstantearekin sarturiko espezieen papera simulazio-ereduan da, espezie hauen harrapaketak arrantza-jarduera murrizteko gai izatea eta arrantza-floten errendimenduan kontutan har daitezkeen bitartean. Baina beren populazio-dinamika simulazio-ereduan sartu gabe, ezinezkoa da kudeaketa planek beren jasangarritasunean duten eragina aztertzea.

Doktorego tesi honetan egindako ikerketa, STEFCk 2015ean (STECF, 2015) Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen urte anitzeko kudeaketa plana ebaluatzeko aplikatu zuen simulazio-eredua hobetzean oinarritzen da. Alde batetik, eredu eragilean Frantziako eta Espainiako ontzien dinamikak sartuko dira, ICESek Bizkaiko Golkoan jarduten duen arrantza-flota mistoen kontsiderazioa emateko erabiltzen duen floten segmentazio berdina erabiliz (ICES, 2022b,e). Bestalde, eredu eragilean, espezie gehiagoren populazio-dinamikak sartuko dira, datu-mugatuko zein datu-ugariko espezieen populazio-dinamikak. Horretarako, arreta berezia jarri beharko da simulazio-ereduan sartu beharreko espezieen aukeraketan eta datu-mugatuak dituzten espezieen populazio-dinamikaren kalkuluan. Espezieen aukeraketa, Osio et al. (2015)k espezieak lehenetsi ahal izateko garatu zuen metodoaren luzapenean oinarrituko da, eta datu-mugatuak dituzten espezieen populazio-dinamiken kalkulua aldiz, SRA ereduan (Huynh et al., 2020a) oinarrituko da. Hobetutako simulazio algoritmoa, FLBEIA simulazio-ereduan (García et al., 2017) aplikatuko da eta Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen egungo kudeaketa estrategien eta kudeaketa estrategia alternatiboen efektua ebaluatuko da jasangarritasun biologikoari dagokionean harrapatutako arrainak portura eramateko obligazioarekin eta gabe. Oro har, lan hau lagungarria izan daiteke Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen kudeaketa estrategien ebaluazio holistikoa egin ahal izateko eta CFPren helburu nagusienetako bat lortzen laguntzeko: hau da, harrapatutako espezie guztien jasangarritasuna bermatzea eta, aldi berean, arrantza-jarduera epe luzera mantentzea.

1.5 Lan-hipotesia eta helburuak

Doktorego tesi honetan egindako ikerketa gidatu duen lan-hipotesia (Dewey, 1938; Shields and Tajalli, 2006) ondoko hau da:

"Bizkaiko Golkoan jarduten duten arrantza-flota demertsal mistoen urte anitzeko kudeaketa plana ebaluatzeko, espezieak egokiro aukeratu behar dira eta datu-mugatuak dituzten espezieen dinamikak era fidagarrian ezagutu behar dira espezie enblematikoenen jasangarritasunari dagokionez kudeaketa estrategiek duten errendimenduaren ebaluazio holistikoa lortzeko".

Lan-hipotesiaren egiatasuna hiru ikerketa-helburu zehatzen bitartez ebaluatu da:

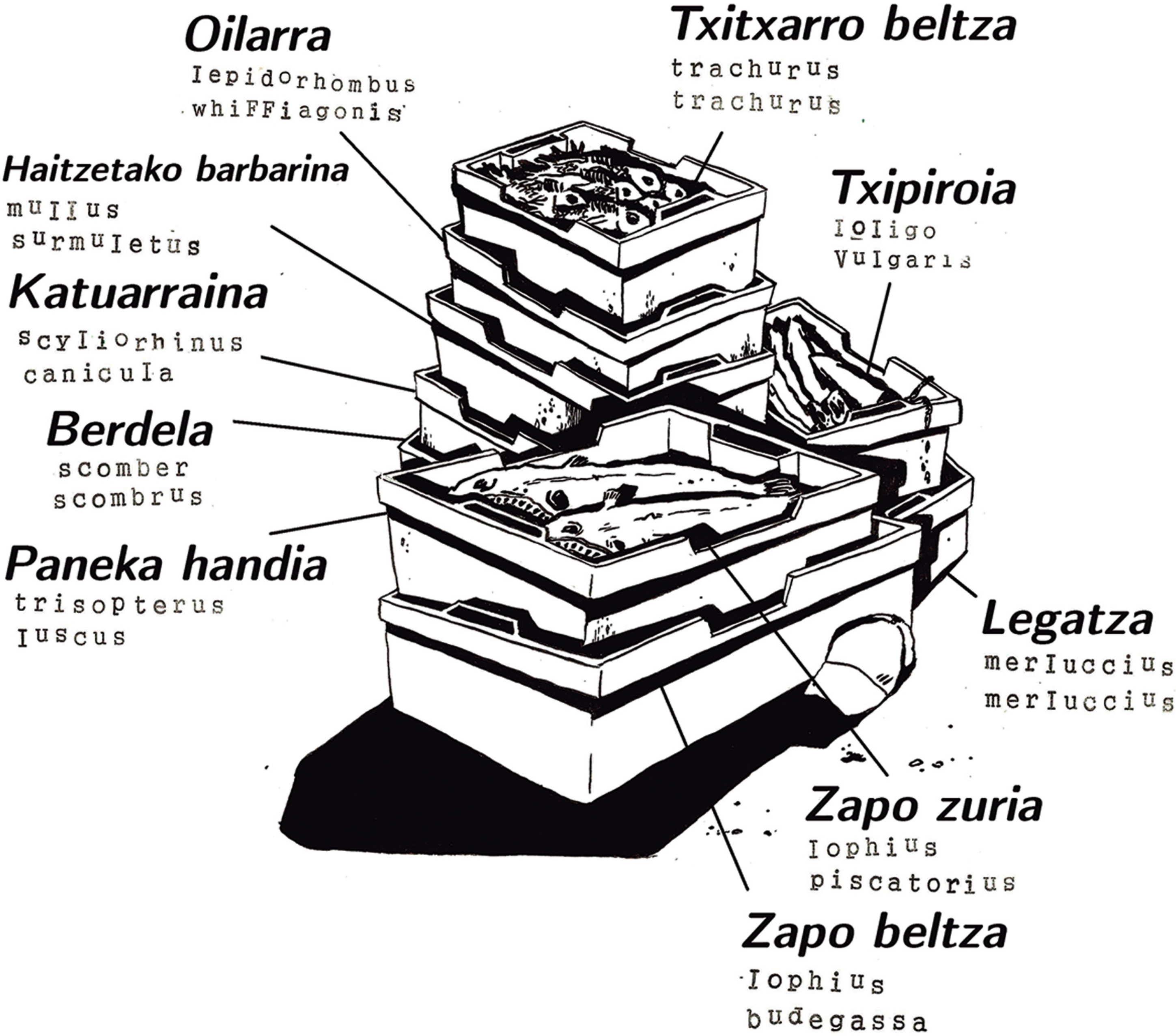
1. Kudeaketa estrategiak ebaluatzeko simulazio-ereduan sartu behar diren espezieak identifikatzea, arrisku-egoera larrienean, ustiapen maila altuenean eta/edo flotarentzat ekonomikoki garrantzitsuenak diren espezieak identifikatuz.

2. "Auto-proba" simulazioaren bidez, datuen eskuragarritasun ezberdineko agertokietan SRA ereduak populazio-dinamiken inguruko kalkulu zehatzak lortzeko duen gaitasuna ebaluatzea.

3. Bizkaiko Golkoan jarduten duten flota demertsal mistoen egungo kudeaketa estrategiek eta kudeaketa estrategia alternatiboek izan dezaketen eragina ebaluatzea, espezie enblematikoenen jasangarritasunari dagokionean.

1.6 Tesiaren egitura

Doktorego tesiaren 1.Kapituluak, sarrera orokor bat eskaintzen du, Europar Batasuneko arrantza-flota mistoak kudeatzeko eta espezie anitzen populazio dinamikak simulazio-ereduetan sartzeko erronka nagusienak azpimarratuz. Ondoren, hiru kapitulu datoz, bakoitzak bere sarrera, materialak eta metodoak, emaitzak eta eztabaida atalak dauzka eta ikerketa-helburu zehatz bat lantzen du. Hasteko, 2.Kapituluan simulazio-ereduan sartu beharko diren espezieak identifikatu dira. Espezieen aukeraketa, Osio et al. (2015)k garatutako espezieen lehentasun metodoaren hedapenean oinarritzen da. Espezieak aukeratzeko oinarria izan da: arrantza-jarduerak espeziean duen eragina PSA bidez kalkulatua (Hobday et al., 2011), harrapaketa-maila eta garrantzi ekonomikoa. 3.Kapituluan, simulazio bidez, SRA ereduaren (Huynh et al., 2020a) erabilera potentziala aztertu da espezieen populazio-dinamiken kalkulu zehatzak lortzeko datu-eskuragarritasun ezberdinetako agertokietan. SRA eredua da 4.Kapituluko simulazio-ereduan datu-mugatuak dituzten espezieen dinamikak sartzeko erabili den eredua. 4.Kapituluak FLBEIA (García et al., 2017) erabiliz sortu den simulazio algorithmoa deskribatzen du, non datu-ugariko espezie guztiak sartu diren beren ebaluazio-eredutik lortutako kalkuluetan oinarrituta, bai eta datu-mugatuak dituzten zenbait espezie ere, SRA eredua oinarri hartuta. Simulazio horrek egungo kudeaketa estrategiak eta kudeaketa estrategia alternatiboek espezieen jasangarritasunean duten errendimendua ebaluatzea ahalbidetu du, portura eramateko obligazioarekin eta gabe. 5.Kapituluan, aurreko hiru kapituluetatik ateratako limitazio nagusienak eta aurkikuntza garrantzitsuenak, baita tesiaren ostean gara daitezkeen balizko ikerketak ere aurkezten dira. 6.Kapituluan, ondorio nagusiak laburbiltzen dira. 7.Kapituluak tesian zehar gauzatutako ekarpen zientifikoen zerrenda jasotzen du. Azkenik, C.Eranskinak, E.Eranskinak eta F.Eranskinak hiru kapitulu nagusietako material osagarria biltzen dute.



2 Kapitulua

MSEn sartu behar diren espezieen identifikazioa

The English version of this chapter is in Appendix B

2.1 Sarrera

Urte anitzeko kudeaketa plana martxan jarri aurretik, kudeaketa planak ezartzen dituen neurriek sisteman duten eragina ebaluatu beharko litzateke, kudeaketa planak bilatzen dituen helburuak neurri horiekin lortu daitezkela ziurtatzeko. Arrantza-flota mistoen kudeaketa planen eragina ebaluatzea baina, erronka handia da: flota horiek espezie ezberdin asko harrapatzen dituzte eta ebaluazio-ariketan sarturiko espezieen kopuruak, ondorioz, nahiko zabala izan behar du flotaren jarduera egiazki islatu ahal izateko. Honela, simulazio-eredua aplikatzeko lehen urratsa, simulazioan sartu behar diren espezieak identifikatzea da, kudeaketa helburuen lorpen-maila ebalua daitekeela bermatzeko.

Arrantza-floten jasangarritasuna ebaluatzeko beharraren ondorioz, arrantzaren eraginez sortzen den arrisku ekologikoa ebaluatzeko metodo ezberdinak garatu ziren (Hobday et al., 2007; MSC, 2001). Metodo horietako bat da PSA (Hobday et al., 2011, 2007). PSA datu-eskari baxuko eredu erdi kuantitatiboa da. PSA, Stobutzki et al. (2001)k garatu zuen lehen aldiz, Australia iparraldean otarrainxka harrapatzen zuten arraste-ontziek baztertzen zituzten espezieen arriskua ebaluatzeko. PSAk zehazten duen arrisku-maila, egoera larrienean dauden espezieak identifikatzeko erabiltzen da, hauen ebaluazio kuantitatiboa lehenesteko ondoren (Hobday et al., 2007). Bere malgutasun eta datu-eskakizun minimoak direla-eta, mundu osoko arrantza-jarduera aztertzeko aplikatu izan da, xede-espezieetatik bazterkinetaraino. Azken horien artean daude itsas hegaztiak, itsas dortokak, marrazoak, arraiak eta itsas ugaztunak (Cortes et al., 2015; Lucena-Fredou et al., 2017; Okemwa et al., 2016; Waugh et al., 2012). Era berean, espezieei aplikatzeaz gain, ekosistemaren beste osagai batzuei ere aplikatu zaie: komunitateei edo habitatei, esaterako (Williams et al., 2011).

Kapitulu honen xedea da euskal arrantza-flota demertsal mistoak harrapatzen dituen espezie guztien arrisku potentziala ebaluatzea, eta euskal arrantza-flota demertsal mistoak gehien ustiatzen dituen, egoera larrienean dauden eta/edo ekonomikoki garrantzitsuenak diren espezieak identifikatzea. Hasteko, flota honek ustiatzen dituen espezieen arrisku ekologikoa kalkulatu da PSA erabiliz (Hobday et al., 2011). Ondoren, espezieak sailkatu dira beren arrisku potentzialaren (PSA), mozkinen eta harrapaketa-bolumenaren arabera, Osio et al. (2015)k proposatu zuen espezieen lehenespen metodoa aldatuz. Azkenik, sailkapen honek arrantza-flota demertsal mistoaren kudeaketa plana ebaluatzeko erabili beharko liratekeen espezieak identifikatzeko izan dezakeen erabilgarritasuna eztabaidatu da.

2.2 Materialak eta metodoak

2.2.1 Espezieak hautatzea

Euskal arrantza-flota demertsala Europar Batasunaren mendebaldeko uretan, Kantauri itsasoaren kostaldetik Eskozia mendebalderaino, jarduten duen flota mistoa da. Ikerketa, baina, Bizkaiko Golkora mugatu da (8abd dibisioak, ICES) (1.7.Irudia), bertan gertatzen baitira flotaren lehorreratze gehienak (lehorreratze guztien %80 inguru 2001 eta 2017 artean). Arrantza-aparailu ezberdinak erabiliz jarduten duten hiru flotek osatzen dute euskal arrantza-flota demertsal mistoa: hondoko arrasteontziak (OTB), bikotekako arraste-ontziak (PTB) eta tretzaontziak (LLS). OTB ontziek 10-1.400 m-ko sakoneretan jarduten dute 70 mm-ko tamaina duten sareekin.

PTB ontziek, 24-400 m-ko sakoneretan, 100 mm-ko sareekin. LLS ontziek 300 m baino gutxiagoko sakoneretan ontzi bakoitzak 1.000 amu dituela.

Flotaren jarduera sei modalitatetan banatzen da, aparailuaren eta xede-espezieen arabera (Iriondo et al., 2010) (2.1.Taula). Arrantza aktibitatearen porturatutako espezieen konposizioa ezagutzeko 2001 eta 2017 bitarteko lehorreratzeen datuak erabili ziren. Bazterkinak zehazteko, 2003 eta 2017 bitartean egindako behaketetatik lorturiko informazioa erabili zen. Guztira, sei modalitateen lehorreratzeek eta bazterkinek 150 espezie ingururen harrapaketak biltzen ditu. Modalitate bakoitzeko lehorreratzeen %95 eta bazterkinen %95 osatzen duten espezieak aukeratu ziren. Halaber, harrapatutako arrain guztiak portura eramateko betebeharraren ondorioz, euskal arrantza-flota demertsal mistoaren jarduera muga dezaketen espezie guztiak sartuak zeudela ziurtatzeko, arrantza kuota zehaztua zuten espezie guztiak ere barnean sartu ziren, nahiz eta beren lehorreratze- eta bazterkin-ekarpenak %95 horren barruan ez egon. Lehorreratzeen eta bazterkinen inguruko informazioa AZTIren arrantzari buruzko datu-basetik atera zen, laginen zein behaketen datuak biltzen baititu.

Taula 2.1: Euskal arrantza-flota demertsal mistoa osatzen duten sei modalitateen deskribapena, arrantza-aparailuaren eta xede-espezieen arabera. Aparailuak: hondoko arraste-ontziak (OTB), bikotekako arraste-ontziak (PTB) eta tretzaontziak (LLS).

Modalitatea	**Aparailua**	**Xede-espeziea**	**Espezieak**
OTB_DEF	OTB	Arrain demertsalak	Legatza, oilarra, oilar txikia, zapo zuria, zapo beltza, paneka handia
OTB_MCF		Zefalopodo eta arrain demertsalak	Txipiroia, begihandia, txibia, txokoa, txibia arrosa, paneka handia, haitzetako barbarina
OTB_MPD		Arrain pelagiko eta demertsalak	Legatza, berdela, txitxarro beltza
OTB_SPF		Arrain pelagiko txikiak	Berdela, txitxarro beltza
PTB_DEF	PTB	Arrain demertsalak	Legatza
LLS_DEF	LLS	Arrain demertsalak	Itsas aingira, legatza

2.2.2 Produktibitate-suszeptibilitate analisia

PSA gauzatu zen aukeratutako espezieetarako. Produktibitatea ezaugarri demografikoek zehazten dute eta espezie bakoitzerako kalkulatu zen (Hobday et al., 2011). Suszeptibilitatea arrantza-flotaren eta espeziearen arteko elkarrekintzak zehazten du. Euskal arrantza-flota demertsal mistoa suszeptibilitate ezberdineko zenbait aparailuk osatzen dutenez, hasteko, espezie eta aparailu (OTB, PTB eta LLS) bakoitzaren suszeptibilitatea ebaluatu zen (Hobday et al., 2011); ondoren, gainjarritako arrantza-jardueren eragin metatua kalkulatu zen, Micheli et al. (2014)k proposatu zuen suszeptibilitate metatuaren metodoa erabiliz. Produktibitate eta suszeptibilitate ezaugarriak (aurrerantzean, atributuak) 1 (arrisku baxua) eta 3 (arrisku altua) artean puntuatu ziren eta puntuaketa hauetaz baliatuz arrisku potentziala ere kalkulatu zen (Hobday et al., 2007). Emaitza horiek x-y grafiko batean (hau da, PSA grafikoan) azaldu ziren.

2.2.2.1 Produktibitatea

Produktibitate-atributuak Hobday et al. (2011)k proposatu zuen PSA oinarri hartuta aukeratu ziren. Hobday et al. (2011)k zazpi produktibitate-atributu erabili zituen: luzera maximoa, adin maximoa, helduaro-luzera, helduaro-adina, emankortasuna, ugaltze-estrategia eta maila trofikoa. Atributu horietatik, guk ez genuen maila trofikoa delako atributua erabili (Duffy and Griffiths, 2017; Hordyk and Carruthers, 2018). Izan ere, aditu batzuk (Hobday et al., 2007; Patrick et al., 2010) maila trofikoaren eta produktibitatearen arteko alderantzizko lotura ezartzen badute ere, gure ikerketan azaltzen ziren zenbait espezierentzat, maila trofikoa ez da produktibitatearen adierazgarri. Esaterako, Zefalopodoek, Sepiidae, Octopodidae eta Ommastrephidae familietako espezieek, maila trofiko altua agertzen dute beren kanibalismoaren ondorioz baina oso produktiboak ere badira.

Produktibitate-atributuak 1 (arrisku baxua edo produktibitate handia) eta 3 (arrisku handia edo produktibitate baxua) bitartean puntuatzen dira. Ugaltze-estrategien atalaseei dagokienez Hobday et al. (2011)en horiek mantendu ziren; eramaile bibiparo eta obobibiparoak (produktibitate baxua), errule demertsalak (produktibitate ertaina) eta barreiapenezko ugaltzaileak (produktibitate handia). Horretarako suposatu zen barreiapenezko ugaltzaileak diren espezieek kume gehiago ekoizteko gaitasuna dutela eta, ondorioz, berreskuratze-gaitasun handiagoa dutela beren populazioa murriztua denean, eramaile bibiparo eta obobibiparoak baino (Stobutzki et al., 2001). Kategorizazio horren oinarria da bizi-historia motzagoa duten espezieek, berreskuratzeko gaitasun handiagoa dutela beren populazioa murrizten denean. Produktibitate handiko espezieak txikiak dira, bizi-ziklo laburrekoak, agudo heltzen direnak helduarora eta ugaltzeko gaitasun handia dutenak (Winemiller and Rose, 1992). Beste atributuei dagokienez bi metodo ezberdin probatu genituen atalaseak zehazteko: espezieak tamaina bereko taldetan banatzen dituen zentil-metodoa (Lucena-Fredou et al., 2017) eta K-mean klusterizazio-metodoa. Azken honek antzekoenak diren espezieak biltzen ditu espezie bakoitzaren balio-atributuaren eta taldekako batez besteko balioaren bariantza aintzat hartuz (Patrick et al., 2009). 2.2.Taulak metodo bakoitza erabiliz atributu bakoitzerako lortu ziren atalaseak jasotzen ditu.

Produktibitate-atributu guztiak berdin haztatu ziren eta produktibitate-atributu guztien emaitzen batez bestekoa kalkulatu zen guztizko produktibitate-emaitza lortzeko (Hobday et al., 2011).

Taula 2.2: Produktibitate-atributuen atalase-balioak Hobday et al. (2011)k proposatutakoa egokitu ostean. Atributu-balioak ondoko kategorietan sailkatu ziren: produktibitate baxua, ertaina eta altua. Erreprodukzio-estrategia: beren gametoak uretan askatzen dituzten arrainak (barreiapenezko ugaltzaileak – kanpoko ernalketa, BS), arrautzak erruten dituzten espezieak (obiparoak, erruleak – barruko ernalketa, DS) eta espezie obobibiparoak eta bibiparoak (eramaile bibiparo eta obobibiparoak – barruko ernalketa, LB).

Produktibitate-atributuak	**Produktibitate baxua (Arrisku altua, puntuazioa =3)**		**Produktibitate ertaina (Arrisku ertaina, puntuazioa =2)**		**Produktibitate altua (Arrisku baxua, puntuazioa =1)**	
	K-mean	Zentil	K-mean	Zentil	K-mean	Zentil
Helduaro-adina (urtea)	>6,66	>5	2,78–6,66	2,5–5	<2,78	<2,5
Adin maximoa (urtea)	>31,8	>20	11,5–31,8	12–20	<11,5	<12
Helduaro-luzera (zm)	>124	>35,4	50,5–124	20–35,4	<50,5	<20
Luzera maximoa (zm)	>163	>90	69,3–163	46–90	<69,3	<46
Emankortasuna (min arraultza/urtea)	<3,7E+5	<4,5E+3	3,7E+5–1,8E+6	4,5E+3–8,8E+4	>1,8E+6	>8,8E+4
Ugaltze-estrategia	LB		DS		BS	

2.2.2.2 Suszeptibilitatea

Arrantza-aparailu indibidualak Suszeptibilitatea Hobday et al. (2011)k proposatu zituen lau suszeptibilitate-atributuetan oinarrituta kalkulatu zen: eskuragarritasun edo gainjartze horizontala flota-banaketaren eta espezie-banaketaren artean, aurkigarritasuna edo gainjartze bertikala flota-banaketaren eta espezie-banaketaren artean, selektibitatea edo arrantza-flotak harrapatzen dituen arrainen luzera-banaketa, eta harrapaketa osteko heriotza-tasa edo harrapatu ostean askatuz gero espezieak bizirauteko duen gaitasuna. 1.mailako atributu horiek 2.mailako atributuak konbinatuz zehazten dira. Eskuragarritasuna 2.mailako hiru atributuren emaitza maximoaren arabera zehazten da: banaketa globala, migratzeko gaitasuna eta espeziearen agregazio-portaera. Aurkigarritasuna 2.mailako bi atributuren emaitza minimoaren arabera zehazten da: arrantza-aparailuak espeziearen helduaroko habitatera iristeko duen gaitasuna eta batimetria-gainjarpena arrantza-flotaren eta espezie-banaketaren artean.

2.mailako atributu bakoitzak 1 (arrisku baxua edo suszeptibilitate baxua) eta 3 (arrisku altua edo suszeptibilitate altua) puntu bitartean jasotzen ditu. Puntuak esleitzeko irizpideak euskal arrantza-flota demertsal mistoaren errealitatera eta honek arrantzatzen dituen espezieen ezaugarrietara egokitu ziren (2.3.Taula). Helduaroko habitateko gainjartzearen atalaseak eta batimetria-gainjarpenaren atalaseak aparailu bakoitzerako zehaztu ziren (C Eranskineko C.1.Taulan ikus daitezke aparailu bakoitzean erabilitako atalaseak). Selektibitate-atalaseak zehazteko, Marine Stewardship Council erakundean oinarritu ginen (Hordyk and Carruthers, 2018; MSC, 2001). Honek, arrisku baxua esleitzen dio harrapaketa-luzera (Lc) heldutasun-luzera (Lm) baino handiagoa denean; arrisku ertaina 0,5Lm <Lc <Lm formula aplikagarria denean eta arrisku handia Lc <0,5Lm formulak errealitatea islatzen duenean. Harrapaketa osteko heriotza-tasa oro har zehaztu da, marrazoek eta arraiek beste espezie batzuek baino gaitasun handiagoa dutela bizirauteko onartuz.

Taula 2.3: Suszeptibilitate-atributuen atalase-balioak Hobday et al. (2011)k proposatutakoa egokitu ostean. Atributu-balioak ondoko kategorietan sailkatu ziren: suszeptibilitate baxua, ertaina eta altua. Eskuragarritasun-atributuaren emaitza, banaketa globala, migrazio gaitasuna eta espeziearen agregazio-portaera atributuen emaitza maximoa izan zen. Aurkigarritasun-atributuaren emaitza, helduaroko habitata eta batimetria atributuen emaitza minimoa izan zen. Aurkigarritasun- eta selektibitate-atributuak aparailu bakoitzerako kalkulatu ziren. Lc, %50 arrantzatzen direnen luzera. Lm, %50 heldu direnen luzera.

Suszeptibilitate-atributuak		**Suszeptibilitate baxua (Arrisku baxua, puntuazioa =1)**	**Suszeptibilitate ertaina (Arrisku ertaina, puntuazioa =2)**	**Suszeptibilitate altua (Arrisku altua, puntuazioa =3)**
Eskuragarritasuna	Banaketa globala	Mundu osoa (W)	Ipar Atlantikoa (NA)	Bizkaiko Golkoa (BoB)
	Migrazioa	Migratzaile handiak	Hedatzeko oztopo gutxi	Bizi zikloa itsasoan amaitzeko gaitasunik ez
	Agregazio-portaera	Elikadura-, ugalketa- eta errute-ohitura berdintsuak urte osoan zehar	Urtaroka aldaketa txikiak baina ugalketa ez dago urtaro zehatz bati lotua	Ugalketa-agregazioak eratzen dira
Aurkigarritasuna	Helduaroko habitata	Gainjartze baxua arrantza-aparailuarekin	Gainjartze ertaina arrantza-aparailuarekin	Gainjartze altua arrantza-aparailuarekin
	Batimetria	Gainjartze baxua arrantza-aparailuarekin	Gainjartze ertaina arrantza-aparailuarekin	Gainjartze altua arrantza-aparailuarekin
Selektibitatea		Lc >Lm	0,5Lm <Lc <Lm	Lc <0,5Lm
Harrapaketa osteko heriotza-tasa		Bizirik askatuta	Askatutako zenbait bizirik	Hilik askatuta

1.mailako suszeptibilitate-atributu guztiak berdin haztatu ziren eta guztizko suszeptibilitatea batez besteko geometriko gisa kalkulatu zen (Hobday et al., 2011). Aparailu batek espezie baten arrainik harrapatu ez zuen kasuetan (C Eraskineko C.3.Taula, C.4.Taula eta C.5.Taula) espezie hori ez harrantzatzeko kanpo faktoreak zeudela onartu zen. Ondorioz, espezie horiek ez dira kontuan hartu dagokion PSA grafikoan.

Arrantza-flota osoa Arrantza-jarduera guztiaren inpaktu metatua kalkulatzeko Micheli et al. (2014)k proposatutako suszeptibilitate metatua (AS) erabili zen:

$$AS = min\left(3, 1 + \sqrt{(S_{OTB} - 1)^2 + (S_{PTB} - 1)^2 + (S_{LLS} - 1)^2}\right), \qquad (2.1)$$

non S_{OTB}, S_{PTB}, eta S_{LLS} OTB, PTB, eta LLS floten suszeptibilitate-puntuazioak diren, hurrenez hurren. Ondorioz, AS adierazlea handiagoa da arrantza-flota kopurua igo ahala eta eragin metatu potentziala flota bakar batek sortutako hori baino handiagoa izan daiteke (Halpern et al., 2008). AS ren balio minimoa eta maximoa 1 eta 3 dira, hurrenez hurren, jatorrizko PSA indizearen suszeptibilitate-puntuen aukerarekin koherentzia mantentzeko (Micheli et al., 2014).

2.2.2.3 Arrisku potentziala

Espezie bakoitzaren arrisku potentziala (R) kalkulatu zen, aparailu bakoitzerako eta arrantza-flota osorako, jatorritik dagoen distantzia euklidear gisa:

$$R = \sqrt{P^2 + S^2}, \qquad (2.2)$$

non P eta S produktibitate- eta suszeptibilitate-puntuazioak diren, hurrenez hurren. Arrisku hori 1,41etik (puntuazio guztiak 1 direnean) 4,24raino doa (puntuazio guztiak 3 direnean). Puntuazio guztiak lortzeko aukerak berdinak direla kontuan hartuz, balioen heren bat 2,64tik behera dago eta balioen beste heren bat 3,18ren gainetik (Hobday et al., 2007). Honela, 2,64ko arriskutik behera dauden espezieak arrisku baxuko gisa, 2,64 eta 3,18 bitartean daudenak arrisku ertaineko gisa eta 3,18ren gainetik daudenak arrisku altuko gisa sailkatzen dira (Hobday et al., 2007).

2.2.2.4 Datuen kalitate-indizea

Datuen kalitate-indizea erabili zen arrisku potentzialari loturiko puntuazioaren ziurgabetasuna neurtzeko (Patrick et al., 2009). Ziurgabetasun horren zioa erabilitako datuen kalitate eskasa edo inolako daturik ez izatea izan daiteke. Patrick et al. (2009)en proposamenari jarraituz, datuen kalitate-indizeak bost maila ditu, daturik onenetik (edo sinesmen handia puntuazioan) datu-absentziaraino (edo sinesmen eskasa puntuazioan) doazenak (2.4.Taula). Hasteko, datuen kalitatearen araberako puntuazio bat esleitu zitzaion produktibitate-atributu eta 2.mailako suszeptibilitate-atributu bakoitzari. 1.mailako suszeptibilitate-atributuen kalitate-indizea 2.mailako suszeptibilitate-atributuen kalitate-indizearen batez besteko haztatu gisa kalkulatu zen. Ondoren, produktibitate, suszeptibilitate eta arrisku potentziala kalkulatzeko erabilitako datuen kalitate-indize orokorrak kalkulatu ziren, atributu nagusi horiek jasotako puntuazioaren batez besteko haztatu gisa. Atributu zehatz baten inguruko daturik ez genuenean, kalitate-puntuazio txarrena esleitu zitzaion (hau da, “5”) eta ez zen kontuan hartu produktibitate- edo suszeptibilitate-puntuaziorako. Datuen kalitate-indizeen puntuazioak eta produktibitate- eta suszeptibilitate-atributu bakoitzaren iturriak C Eranskineko C.6.Taulan eta C.7.Taulan jasota daude. Patrick et al.

(2009)ek egin bezala, datuen kalitate-indizea hiru kategoriatan banatu zen –kaskarra >3,5, nahikoa 2,0-3,5; eta ona <2,0 –, arrisku potentzial orokorraren puntuazioaren ziurgabetasuna interpretatzen laguntzeko.

Taula 2.4: Espezie zehatz baten produktibitatearen eta suszeptibilitatearen ziurgabetasuna ebaluatzeko erabili ziren datuen kalitate-indizearen bost mailak. Patrick et al. (2009)en proposamena egokituta.

Maila	Deskribapena	Adibidea
1	**Daturik onenak.** Informazioa interesatzen zaigun espezieari eta areari buruzko datu aproposak eta esanguratsuak bilduz osatu da.	Espeziearen ebaluazioa datu esanguratsuak aintzat hartuz egin da, argitaratutako literatura biltzeko metodoak aplikatu dira.
2	**Datu egokiak.** Informazioa osatzeko estaldura mugatuko edo osotasunean egiaztatu ezin izan diren datuak edo beste edozein arrazoirengatik 1.mailako datuak bezain onak ez diren datuak erabili dira.	Denbora edo espazio-irismen mugatuko datuak, zaharkitua egon daitekeen informazioa, etab.
3	**Datu-mugatuak.** Bariazio handiko eta konfiantza mugatuko kalkuluak, taxon edo bizi-estrategia berdintsuan oinarrituta egon daitezkeenak.	Antzeko genero edo familia, etab.
4	**Datu oso mugatuak.** Informazioa adituen iritzian edo literatura orokorreko artikuluetan oinarritua dago.	Erreferentziarik gabeko datu orokorrak.
5	**Daturik ez.** Inolako daturik ez izateagatik ezin bada ezta iritzi aditu bat ere erabili, datuen kalitate-indizearen atributu honi 5 puntu esleitzen zaizkio eta ez da kontuan hartzen produktibitate- edo suszeptibilitate-puntuazioa zehazteko. Grafiko batean jasotzean, produktibitate- edo suszeptibilitate-puntuazioa zehazteko atributu bat gutxiago aplikatuko da eta gertaera hori eta horren ondoriozko ziurgabetasuna nabarmendu egingo da, datuen kalitate-indizearen bitartez.	

2.2.2.5 Produktibitate-atributuen erredundantzia

Produktibitate-atributu batzuk (esaterako, luzera maximoa, adin maximoa, helduaro-luzera eta helduaro-adina) korrelazioan egon daitezkeenez, sentiberatasun-analisi bat egin zen atributu horietako baten bat PSAn erredundantea izan daitekeen ala ez ikusteko. Hasteko, produktibitate-atributu biren arteko korrelazioa ebaluatzeko erregresio lineala erabili zen (R2 >0,5 duten atributu pareak korrelazio handia zutela erabaki zen). Jarraian, korrelazioan zeuden atributuen erredundantzia aztertu zen, korrelazio handian zeuden bi atributuetako bat produktibitate orokorraren kalkulutik eta datuen kalitate-mailaren kalkulutik kenduz. Horrez gain, korrelazioan zeuden bi atributuetako datuen kalitatea aztertu zen. Analisi horrek, korrelazioan zeuden atributuren bat produktibitate orokorra kalkulatzeko ariketatik kentzea edo mantentzea egokiagoa zen erabakitzen lagundu dezake. Adibidez, korrelazioan dauden bi atributuetako bati buruzko datuen kalitatea kaskarra bada eta produktibitatearen puntuazioa asko aldatzen bada atributu hau kontuan hartu hala ez, atributu hori kenduz gero produktibitatearen kalkulua zehatzagoa izango da.

2.2.2.6 Alderatzea IUCN eta ICES kategoriekin

PSA bidez kalkulaturiko arrisku potentzialaren puntuazioak, IUCN (International Union for Conservation of Nature) erakundearen lista gorriarekin (IUCN, 2019) alderatu ziren. Sailkapen horrek ondoko galtze-kategoriak hartzen ditu aintzat: ebaluatu gabe (NE), datu kaskarrak (DD), kezka-maila baxua (LC), mehatxupean erortzear (NT), egoera zaurgarrian (VU), arriskuan (EN) eta arrisku larrian (CR). Eskuragarri zegoenean, Europako kategoria erabili zen. Kategoria globala, bestela. ICES kategoriak ere aztertu genituen (ICES, 2012). Honek, espezie populazioak datuen eskuragarritasunaren arabera sailkatzen ditu (1.1.Taula).

2.2.3 Espezieen sailkapena

Espezieak lehenesteko metodoaren helburua, maila handian ustiatzen diren, egoera zaurgarrian egon daitezkeen eta/edo arrantza-flotarako ekonomikoki garrantzitsuak diren espezieak identifikatzea zen. Espezieak lehenesteko metodo honek, Osio et al. (2015)k proposatutako metodoa du oinarri. Metodo horrek lehorreratzeen bolumena, batez besteko prezioa eta arrisku potentzialaren puntuazioa hartzen ditu aintzat espezieak lehenesteko. Gure kasuan, bazterkinak ere kontuan hartu genituen espezieak lehenesteko, baztertzen diren espezie batzuen bolumena handia baita arrantza-flota honen kasuan. Honela, Osio et al. (2015)k proposatutako metodoa zabaldu egin genuen. Gure kasuan mozkinak (€), harrapaketen bolumena (Kg) eta arrantza-flotaren arrisku potentzial orokorraren puntuazioa (PSA) erabili ziren espezieak lehenesteko edo sailkatzeko. Horrez gain, espezie bakoitzak sailkapenean zuen posizioa zehazteko hiru aldagai hauek (mozkin, harrapaketa eta arrisku) lotzeko aplikatu genuen metodoa ez zen Osio et al. (2015)ren berdina izan. Gure azterlanean, espezie multzo txiki baten harrapaketak, neurrigabeki altuak ziren analisian sarturiko espezie gehienekin alderatuta. Eta horrek, kopuru txikiagoan harrapatzen ziren espezieen artean zegoen aldakortasuna ezkutatzen zuen. Honela, sailkapena zehazteko irizpidea aldagaien arteko biderketa izan ez gero Osio et al. (2015)n bezala, datuak 0 eta 1 bitartean estandarizatuta beren balioekin, espezie bakoitzaren posizioa sailkapenean, arrisku-mailak zehaztuko zuen gehienbat, lehorreratze- edo baztertze-maila handiak zituzten kasu gutxi batzuren kasuan ezik. Horrenbestez, aldagai bakoitzak sailkapen-posizioari egingo zion ekarpena orekatzeko, espezieak aldagai bakoitzaren (mozkin, harrapaketa eta arrisku) arabera sailkatu ziren, bakarka, eta sailkapeneko azken posizioa zehazteko, sailkapen horietako bakoitzean espezieak zuen posizioa biderkatu zen. Honela, era orokorrean, sailkapeneko lehen postuetan dauden espezieek ekarpen handiagoa egiten diete mozkinei eta/edo maila handiagoan ustiatzen dira eta/edo arrisku-egoera handiagoan daude. Sailkapeneko azken postuetan dauden espezieak, ordea, ekarpen txikiagoa egiten diete mozkinei eta/edo maila baxuagoan ustiatzen dira eta/edo arrisku-egoera baxuagoan daude.

Mozkin guztiak kalkulatzeko, kilogramoko batez besteko prezioa eta lehorreratutako kilogramoak biderkatu ziren. Espezie bakoitzaren batez besteko prezioa (€/kg) 2008 eta 2015 urteen artean kalkulatu zen, Europar Batasunaren Urteko Ekonomia Txosteneko datuak oinarri hartuta (STECF, 2017). Arrantza-flotaren lehorreratzeen eta bazterkinen datu historikoak, AZTIren datu-basetik atera ziren, 2003-2017 epealdirako. Lehorreratzeen datuak, portuetan harturiko laginen inguruko informazioak batuz lortu ziren. Bazterkinen inguruko datuetarako aldiz, behaketen datuak estrapolatu ziren, neurtutako laginak eta arrantza-flota osoa aintzat hartuz. Azkenik, mozkinak eta harrapaketen bolumenak kalkulatzeko erabili ziren epealdiak sailkapenaren-emaitzetan zuten sentiberatasuna aztertu zen (azken 10, 5 eta 3 urteetako datuak).

2.3 Emaitzak

2.3.1 Espezieak hautatzea

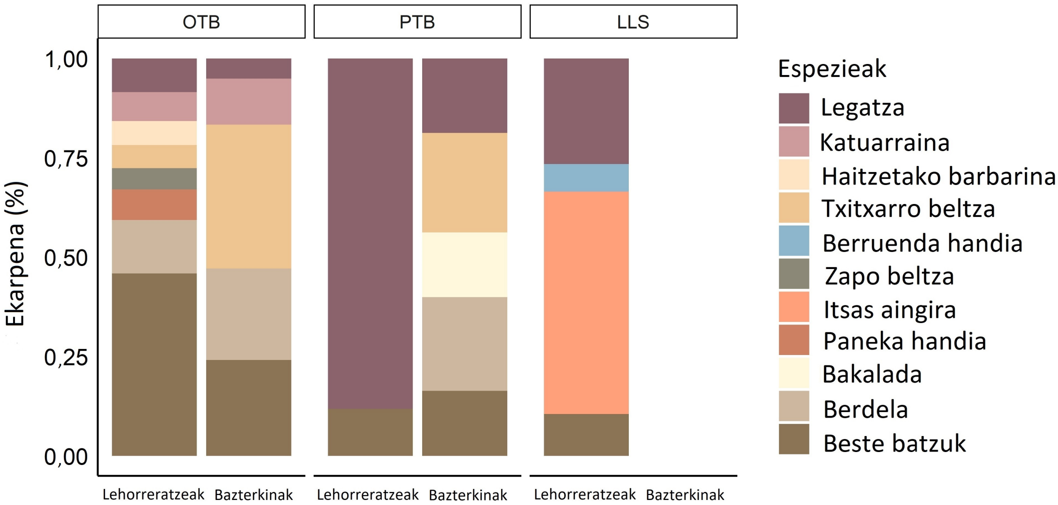
64 espezie, modalitate ezberdinen lehorreratze eta bazterkinen %95aren erantzuleak ziren. Euskal arrantza-flota demertsal mistorako kuota zehaztuta duten hiru espezie (zigala, platuxa leuna eta mielga) ez zeuden %95 horren barruan baina analisian sartu ziren. Horrenbestez, 67 espezie hautatu genituen guztira (2.5.Taula).

Taula 2.5: Hautatutako espezieen izen zientifikoa eta izen arrunta.

Izen zientifikoa	Izen arrunta
Argentina silus	Abixoi handia
Argentina sphyraena	Abixoia
Callionymus lyra	Eiherazain handia
Cancer pagurus	Buia
Capros aper	Karnabalitoa
Cepola rubescens	Xingolarraina
Chelidonichthys cuculus	Perloi kukua
Chelidonichthys lucerna	Perloi handia
Chelidonichthys obscurus	Perloi iluna
Conger conger	Itsas aingira
Dicentrarchus labrax	Lupia arrunta
Dicologlossa cuneata	Mihi-arrain buruhandia
Dipturus batis	Arraia grisa
Dipturus oxyrinchus	Moko-arraia
Eledone cirrhosa	Olagarro zuria
Engraulis encrasicolus	Antxoa
Eutrigla gurnardus	Perloi beltza
Galeorhinus galeus	Gelba
Helicolenus dactylopterus	Kabrarroka
Illex coindetii	Pota
Lepidorhombus boscii	Oilar txikia
Lepidorhombus whiffiagonis	Oilarra
Lepidotrigla cavillone	Perloi latza
Leucoraja naevus	Arraia zerra
Loligo forbesii	Begihandia
Loligo vulgaris	Txipiroia
Lophius budegassa	Zapo beltza
Lophius piscatorius	Zapo zuria
Melanogrammus aeglefinus	Eglefinoa
Merlangius merlangus	Liba
Merluccius merluccius	Legatza
Microchirus variegatus	Mihi-arrain pintoa
Micromesistius poutassou	Bakalada
Molva molva	Berruenda handia

Izen zientifikoa	Izen arrunta
Mugil cephalus	Korrokoi bizkarbeltza
Mullus surmuletus	Haitzetako barbarina
Mustelus asterias	Toil izarduna
Mustelus mustelus	Toil lisoa
Nephrops norvegicus	Zigala
Octopus vulgaris	Olagarro arrunta
Parastichopus regalis	Espardeina
Pegusa lascaris	Hareako mihi-arraina
Phycis blennoides	Lohitako lotxa
Pleuronectes platessa	Platuxa leuna
Pollachius pollachius	Abadira
Raja clavata	Arraia gastaka
Raja fullonica	Arraia jorralea
Raja montagui	Arraia pikarta
Raja undulata	Mosaiko arraia
Sardina pilchardus	Sardina
Scomber colias	Makaela
Scomber scombrus	Berdela
Scyliorhinus canicula	Katuarraina
Sepia elegans	Txokoa
Sepia officinalis	Txibia
Sepia orbignyana	Txibia arrosa
Solea solea	Mihi-arraina
Squalus acanthias	Mielga
Todarodes sagittatus	Pota beltza
Todaropsis eblanae	Itsasertzeko pota
Trachinus draco	Xabiroi zuria
Trachurus mediterraneus	Txitxarro zuria
Trachurus trachurus	Txitxarro beltza
Trigloporus lastoviza	Perloi zirrindatua
Trisopterus luscus	Paneka handia
Trisopterus minutus	Paneka txikia
Zeus faber	Muxumartina

Hautatutako espezie gehienen ekarpen indibiduala arrantza-aparailuko lehorreratzeei eta bazterkinei, %5etik beherakoa izan zen (2.1.Irudia). PTB flotan lehorreratzeen %88 legatzari zegokion. LLS flotaren kasuan aldiz, lehorreratzeen %56 itsas aingirari zegokion. Itsas aingirari zegokion porzentaiari legatza eta berruenda handiaren ehunekoak gehituz gero, LLS flotaren lehorreratzeen %89 batzen dute. OTB flotaren kasuan, ondoko espezie bakoitzaren ekarpen indibiduala lehorreratze guztiaren %5etik gorakoa izan zen eta lehorreratze guztien %54 suposatu zutelarik: legatza, katuarraina, haitzetako barbarina, txitxarro beltza, zapo beltza, paneka handia eta berdela. Bazterkinei dagokienez, legatzak, txitxarro beltzak eta berdelak %67 eta %64 suposatu zuten, hurrenez hurren, PTB eta OTB floten kasuan. Hiru espezie hauez gain, bakaladak PTB flotaren bazterkinei eta katuarrainak OTB flotaren bazterkinei egin zien ekarpena, lau espezie horiek kontuan hartuta bazterkinen %84 eta %76 batu zuten, hurrenez hurren (2.1.Irudia). Ikusi C Eranskineko C.1.Irudia, C.2.Irudia eta C.3.Irudia espezieek lehorreratzeei eta bazterkinei azken 10, 5 eta 3 urteotan egin dieten ekarpena ikusteko, arrantza-aparailu bakoitzerako.

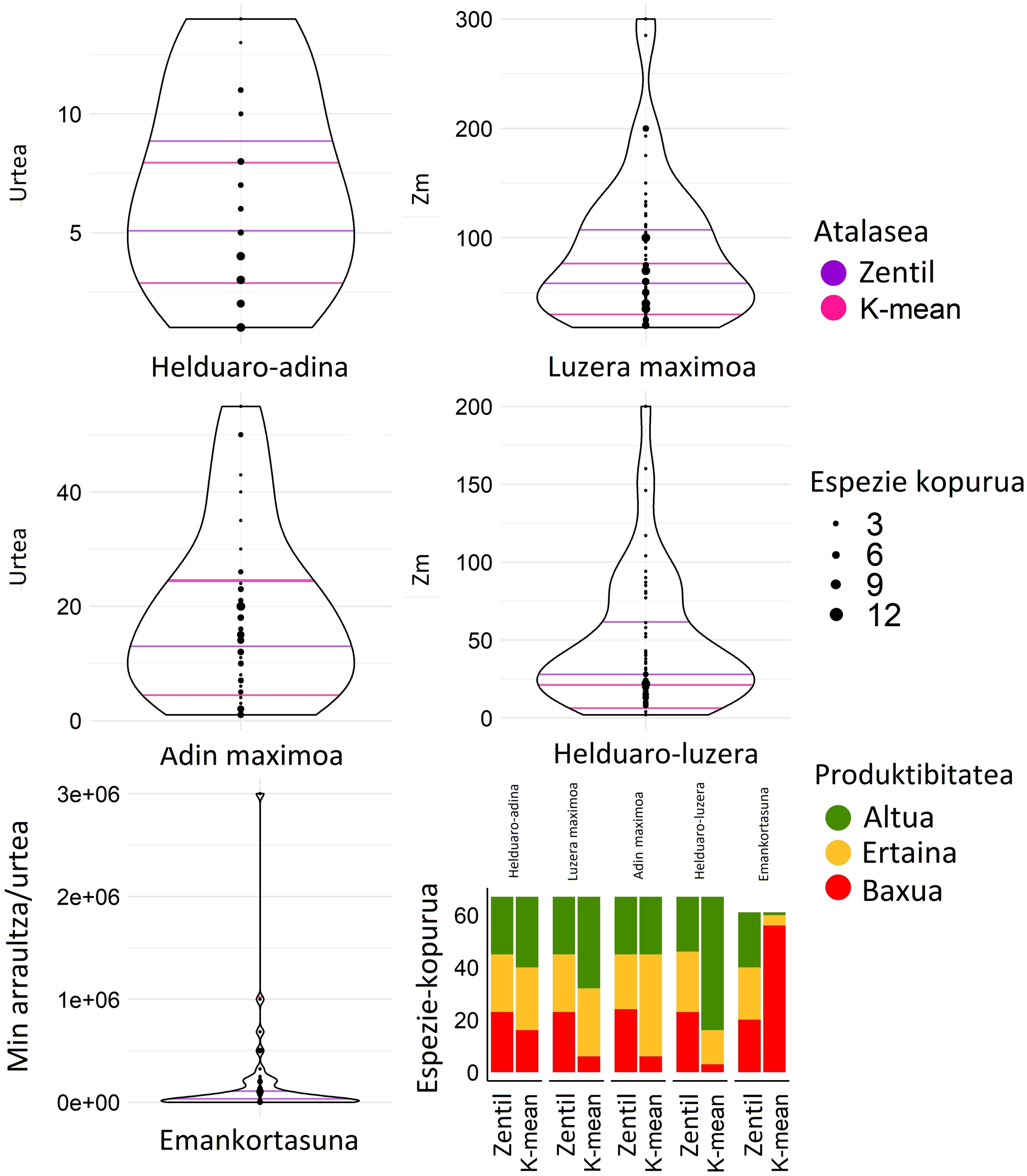


Irudia 2.1: Espezieen ekarpena (%) lehorreratzeei eta bazterkinei, aparailuko: hondoko arraste-ontziak (OTB), bikotekako arraste-ontziak (PTB) eta tretzaontziak (LLS). Beren ekarpen indibiduala %5etik behera duten espezie guztiak "Beste batzuk" atalean bildu dira.

2.3.2 Produktibitate-suszeptibilitate analisia

2.3.2.1 Produktibitatea

Produktibitate-atributuen atalasea definitzeko erabili ziren bi metodoek, zentil eta K-mean metodoek, antzeko atalaseak agertu zituzten helduaro-adina atributurako (2.2.Irudia). Beste atributuen kasuan aldiz, emankortasun-atributurako gehienbat, atalaseak nahiko ezberdinak ziren erabilitako metodoaren arabehera, eta espezieak, ondorioz, produktibitate-kategoria ezberdinetan sailkatu ziren (baxua, ertaina, altua) (2.2.Irudia). Espezie gehienen atributu-balioen arteko ezberdintasuna urria da. Ondorioz, K-mean klusterizazio-metodoa erabiltzean, atalaseek muturreko balioak dituzten espezieen arteko ezberdintasuna soilik islatzen du, espezie gehienen arteko produktibitate ezberdintasunak ezkutatuz (2.2.Irudia). Espezieen arteko produktibitate ezberdintasuna identifikatzeko, K-mean klusterizazio-metodoak atalase desegokiak ematen zituela ikusi genuen. Horrenbestez, zentil-metodoa aukeratu genuen produktibitatearen atributu-atalaseak zehazteko, bereizte handiagoa lortzeko espezie guztien produktibitate-atributuen artean. Ikusi C Eranskineko C.2.Taulan produktibitate-atributuen balioak eta zentil eta K-mean metodoak erabiliz kalkulatutako puntuazioak.



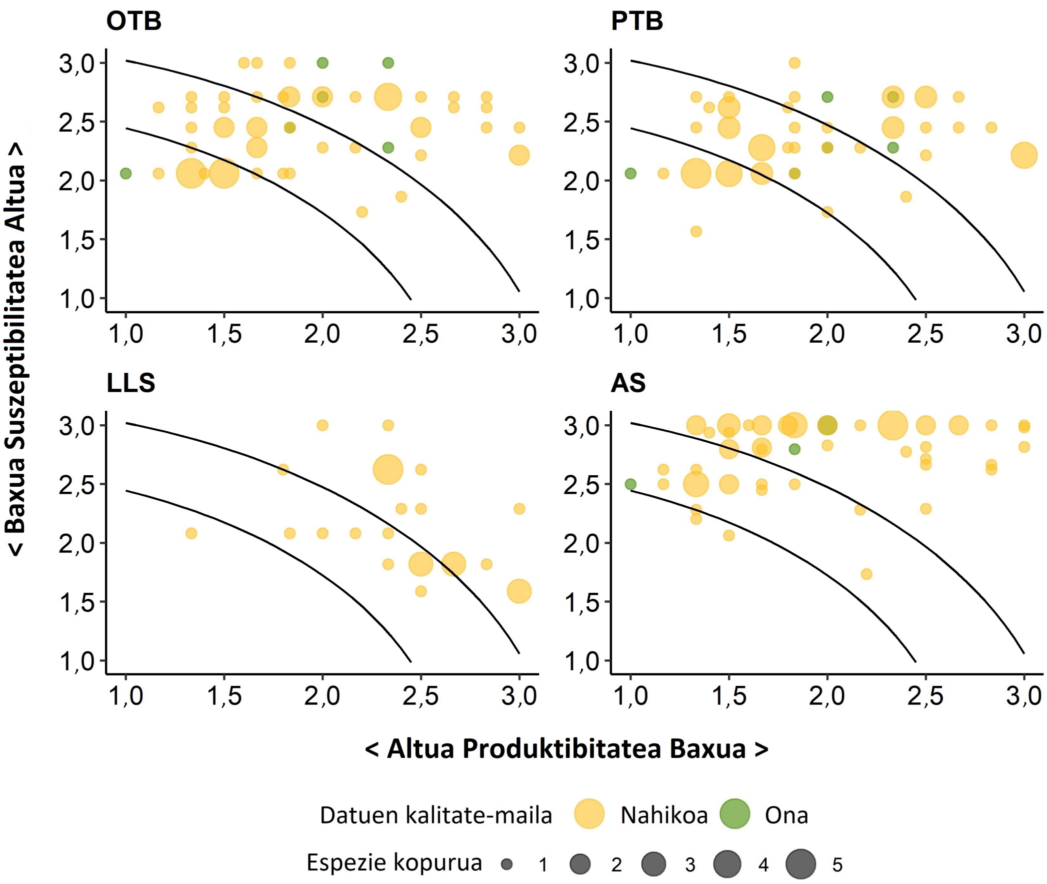
Irudia 2.2: Produktibitate-atributuen (helduaro-adina, luzera maximoa, adin maximoa, helduaro-luzera eta emankortasuna) atalasea metodo ezberdinak erabiliz (zentil eta K-mean). Biolin-itxurako grafikoak espeziearen balio-banaketa erakusten du atributu bakoitzerako; eta atalasea metodoaren araberakoa da (zentil eta K-mean). Barra-grafikoak produktibitate-maila (Altua, Ertaina, Baxua) eta atributu bakoitzean dagoen espezie kopurua islatzen ditu.

2.3.2.2 Aparailu bakarreko arrisku potentziala

Ez zen harrapaketarik egon hautatutako bi espezieri lotuta OTB flotaren kasuan; 13 espezieri lotuta PTB flotaren kasuan; eta 43 espezieri lotuta LLS flotaren kasuan. Flota bakoitzean harrapaketarik izan ez zuten espezieak kendu eta gero, OTB arrantza-flotaren kasuan arrisku altuko 30, arrisku ertaineko 22 eta arrisku baxuko 13 espezie zeudela ikusi genuen (2.3.Irudia). PTB arrantza-flotaren kasuan aldiz, arrisku altuko 19, arrisku ertaineko 25 eta arrisku baxuko 10 espezie zeuden (2.3.Irudia). LLS arrantza-flotaren kasuan, arrisku altuko 14, arrisku ertaineko 9 eta arrisku baxuko espezie 1 identifikatu zen (2.3.Irudia). Hiru arrantza-flota kasuetan, ondoko espezieak arrisku altuan zeuden: itsas aingira, gelba, kabrarroka, zapo beltza, zapo zuria, legatza, toil lisoa, arraia gastaka, arraia jorralea eta mielga (C Eraskineko C.8.Taula). Ikusi C Eranskineko C.3.Taulan, C.4.Taulan eta C.5.Taulan PSAn erabilitako balioak eta puntuazioak.

2.3.2.3 Arrantza-flota osoaren arrisku potentziala

Flota osoaren (AS) inpaktu metatua aztertu zenean, arrisku altuan zeuden espezieen kopuruak gora egin zuen (2.3.Irudia). Arrantza-aparailu edo arrantza-flota guztiak aintzat hartuz, arrisku altuko 47, arrisku ertaineko 18 eta arrisku baxuko 2 espezie zeuden (2.3.Irudia). Gehienbat, marrazoek eta arraiek agertu zituzten arrisku-maila altuak: arraia grisa, moko-arraia, gelba, arraia zerra, toil izarduna, toil lisoa, arraia gastaka, arraia jorralea, arraia pikarta, mosaiko arraia, mielga eta katuarraina (C Eraskineko C.8.Taula).



Irudia 2.3: Produktibitate-suszeptibilitate analisia (PSA). Hondoko arraste-ontzien (OTB), bikotekako arraste-ontzien (PTB), tretzaontzien (LLS) eta arrantza-flota osoaren (AS) grafikoak. Lerro makurrek grafikoa herenetan banatzen dute, arrisku baxua, ertaina eta altua banatuz eta arrisku-maila berdintsuko multzoak islatuz. Datu bakoitzaren kalitate-indizea koloreen bitartez adierazi da eta puntuaren tamainak datuen kalitate-indizea eta produktibitate- eta suszeptibilitate-balio bereko espezieen kopurua islatzen du.

2.3.2.4 Datuen kalitate-indizea

Espezie guztien kasuan, arrisku potentzialari buruzko datuen kalitate-maila nahikoa edo ona zen (2.3.Irudia). Oro har, produktibitate-atributuen datu kalitate-maila ona zen espezie gehienentzat. Ondokoen kasuan, ordea, ez genuen daturik beren emankortasunaren inguruan: karnabalitoa, xingolarraina, olagarro zuria, berruenda handia, korrokoi bizkarbeltza eta espardeina. Horrenbestez, datu kalitate-mailarik baxuena (hau da, "5") esleitu zitzaien espezie hauei atributu honetarako (C Eraskineko C.6.Taula). Selektibitatearen atributurako, ez zegoen daturik espezie guztientzat tretzaontzien kasurako. Horrenbestez, kasu honetan, datuen kalitate-maila 5ekoa izan zen espezie guztientzat (C Eraskineko C.7.Taula).

2.3.2.5 Produktibitate-atributuen erredundantzia

Bi produktibitate-atributu parek korrelazio handia (R2 >0,5) agertu zuten: helduaro-adina eta helduaro-luzera; eta luzera maximoa eta helduaro-luzera (C Eraskineko C.4.Irudia). Atributu horien erredundantzia aztertzeko, korrelazio handia zuten bi atributuetako bat kendu zen eta kasu bakoitzean lortutako produktibitate osoaren eta datu kalitate-mailaren puntuazioak alderatu ziren (2.4.Irudia). Horrez gain, helduaro-adina eta luzera maximoa atributuek helduaro-luzera atributuarekin korrelazio lineala agertu zutenez, helduaro-adina eta luzera maximoa atributuak kenduta, produktibitate osoaren eta datu kalitate-mailaren puntuazioak ere kalkulatu genituen (2.4.Irudia). Kasu guztietan, produktibitate osoaren puntuazioak berdinak ziren (2.4.Irudia). Aldaketa nagusienak ikusi ziren begihandia, espardeina, berruenda handia, pota, muxumartina, zapo zuria, lupia arrunta eta itsas aingira espezieen kasuan; eta aldaketa xumeagoak antxoa, gelba, toil lisoa, mihi-arraina, mielga eta txitxarro beltza espezieen kasuan (2.4.Irudia). Kasu guztietan, produktibitate osoaren datu kalitate-maila ona edo nahikoa izan zen perloi ilunaren kasuan luzera maximoa atributua aintzat hartu ez zenean izan ezik. Kasu honetan, produktibitate osoaren datu kalitate-maila kaskarra izan zen (2.4.Irudia). Korrelazioan zeuden atributuen datu kalitate-maila antzekoa zen (C Eranskineko C.6.Taula).

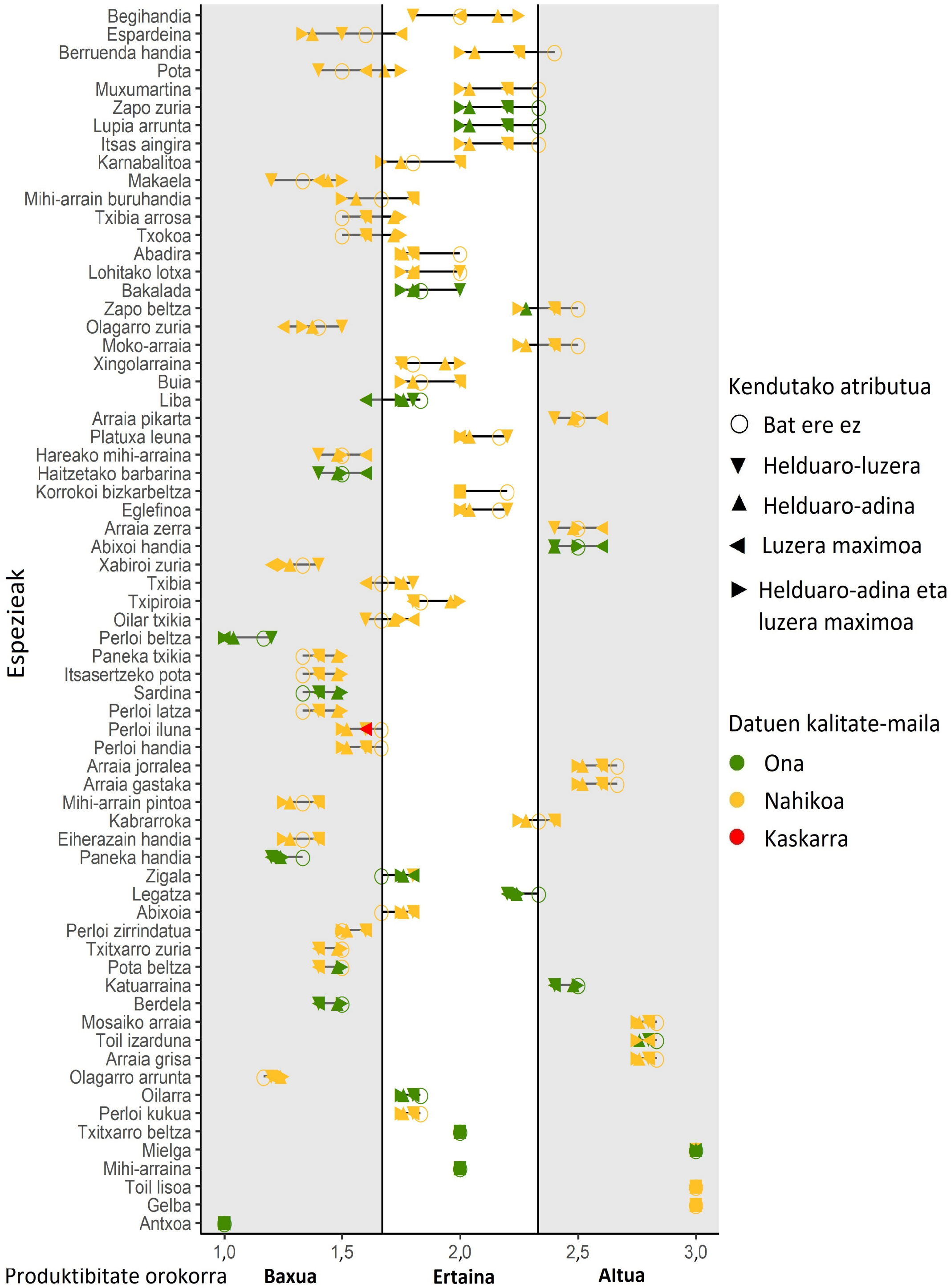
2.3.3 Espezieen sailkapena

Espezieek sailkapenean lortu zuten posizioa ondoko sailkapen indibidualetan zuten posizioen biderketa izan zen: mozkina (€), harrapaketen bolumena (Kg) eta arriskua (PSA puntuazioa) (2.6.Taula). Legatzak lortu zuen azken sailkapenean lehen tokia. Bere prezioa eta bazterkinen bolumena altuak ez baziren ere beste espezieenekin alderatuta, lehorreratze guztien %50a gainditzen zuen eta, horrenbestez, lehen posizioan zegoen mozkinen eta harrapaketen sailkapenetan. Halaber, legatzak arrisku altua agertzen zuen: 9. posizioan zegoen arriskuaren sailkapenean (2.6.Taula). Gelba arriskuaren sailkapenean lehen tokian zegoen baina bere harrapaketa- eta mozkin-bolumen baxuen ondorioz, 12. posizioa eskuratu zuen azken sailkapenean (2.6.Taula). Zapo beltza, haitzetako barbarina, txipiroia, zapo zuria, oilarra eta paneka handia azken sailkapeneko 2., 3., 5., 6., 9. eta 10. posizioan kokatu ziren, hurrenez hurren (2.6.Taula). Espezie hauek, lehen posizioetan zeuden mozkinen sailkapenean beren prezio altuagatik eta lehorreratze-bolumen handiagatik. Halaber, beren bazterkinak oso baxuak ziren (%0,1etik behera guztiak, paneka handia ezik, azken honena %0,3 ingurukoa baitzen). Arrisku-maila altua zuten guztiek, zapo beltzak eta zapo zuriak bereziki (2.6.Taula). Katuarraina, txitxarro beltza eta berdela azken sailkapeneko 4., 7. eta 8. posizioan zeuden (2.6.Taula). Hauek, beren bazterkin-bolumen handiaren eskutik, harrapaketen sailkapeneko lehen postuetan zeuden. Lehorreratze-bolumen handienak zituzten espezieen artean zeuden baita ere, baina beren prezioa baxua zen (euro batetik behera/Kg). Hala ere, mozkinen sailkapeneko goiko erdian kokatuta zeuden. Katuarraina eta txitxarro beltza arrisku altuko espezie gisa sailkatuta zeuden: 6. eta 15. posizioan zeuden arriskuaren sailkapenean, hurrenez hurren (2.6.Taula). Berdelaren arriskua ertaina zen: 29. posizioan zegoen arriskuaren sailkapenean (2.6.Taula).

Harrapaketetan, epealdi ezberdinetan egondako aldaketek ez zuen eraginik izan espezieen sailkapenean (2.1.Irudia, eta C Eranskineko C.1.Irudia, C.2.Irudia eta C.3.Irudia). Sailkapeneko lehen hamar espezieek ez zuten apenas aldaketarik izan aztertutako epealdi ezberdinetan. Salbuespenak ondokoak izan ziren: haitzetako barbarina, txitxarro beltza, berdela eta oilarra. Lau espezie hauen posizioak 5 eta 10 posizio bitarteko aldaketak izan zituzten sailkapenean harrapaketen eta mozkinen epealdi ezberdinak erabili zirenean. Olagarro arrunta izan zen analisian aztertutako epealdi ezberdinen arabera alderik handienak agertu zituen espeziea (2.5.Irudia). Azken urteetako datuak bakarrik aintzat hartuz lortutako sailkapenetan (2008–2017, 2013–2017, 2015–2017 epealdiak), 30 posizio egiten zuen behera epealdi osoa (2003–2017 epealdia) erabilita lortu zen sailkapenarekin alderatuta.

IUCN erakundearen galtze-kategoriaren arabera, desagertzeko arriskuan dauden espezie guztiek (VU, EN, NT edo CR) arrisku-maila altua agertzen zuten PSAn sardina ezik: arrisku ertaineko gisa sailkatua dago PSAn eta NT gisa IUCNren sailkapenean (2.6.Taula). Azken sailkapeneko lehen hamar espezieak IUCNren arabehera, kezka-maila baxu gisa sailkatuta daude, haitzetako barbarina eta txipiroia ezik, hauek datu kaskarrak kategorian daude. ICES kategoriei dagokienez, ikerlanean aztertutako 33 espezie ez daude ICES sailkapenean. Hauetako espezie gehienak lehen hamar posizioetatik at zeuden, txipiroia eta paneka handia kenduta (2.6.Taula). ICES (2012)ek 3. kategorian edo gorago sailkatzen dituen espezieak datu-mugatuak dituzten espezie gisa identifikatzen ditu. Lehen hamar espezieak 1. kategorian daude, katuarraina eta haitzetako barbarina ezik; hauek, 3. eta 5. kategorian daude, hurrenez hurren.

Azkenik, euskal arrantza-flota demertsal mistoak harrapatzen dituen espezieetatik 27 baino ez daude Europako TAC eta kuota-sisteman. Eta horietako sei lehen hamar posizioetan zeuden (2.6.Taula).



Irudia 2.4: Produktibitate-atributuen erredundantziaren analisia, linealki korrelazionatuta dauden atributuak kenduz. Produktibitate orokorraren puntuazioa eta maila (Baxua, Ertaina, Altua) espezie bakoitzerako, produktibitate-atributu guztiak erabiliz (helduaro-adina, adin maximoa, helduaro-luzera, luzera maximoa, emankortasuna eta ugaltze-estrategia), helduaro-luzera atributua kenduz, helduaro-adina atributua kenduz, luzera maximoa atributua kenduz eta helduaro-adina eta luzera maximoa atributuak kenduz. Espezieak kasu ezberdinetan agertu dituzten produktibitate-puntuazioei dagokienean ordenatuta daude, ezberdintasun handiak-ezberdintasunik ez ordenari jarraituz. Datuen kalitate-maila (Ona, Nahikoa, Kaskarra) kasu bakoitzeko produktibitate osoari lotuta dago.

Taula 2.6: Azken sailkapenaren laburpena. Azken sailkapeneko posizioa ondoko sailkapen-posizioen (R) biderketa da: mozkinak [lehorreratzeak (2003–2017 Kg) * prezioa (€/kg)], harrapaketak [2003–2017 Kg] eta arrisku potentzialaren puntuazioa [flota osoaren arriskua]. Mozkinen balio absolutua eta erlatiboa (aztertutako epealdi osoarekiko suposatzen duen ehunekoa), harrapaketak (lehorreratzeak L, gehi bazterkinak B) eta arriskuaren puntuazioa eta maila (baxua, ertaina, altua) erakusten dira azken sailkapeneko posizioak hobeto interpretatu ahal izateko. ICES kategoriak (I), datuen eskuragarritasunean oinarritua [1etik 6raino eta aztertu gabea (N)], IUCN kontserbazio-egoerak (IU) [ebaluatu gabe (NE), datu kaskarrak (DD), kezka-maila baxua (LC), mehatxupean erortzear (NT), egoera zaurgarrian (VU), arriskuan (EN) eta arrisku larrian (CR)] eta espezieak gehieneko kuota ezarrita ote duten ala ez euskal arrantza-flota demertsal mistorako (C). IUCN* ikurrak esan nahi du kategoria hori ebaluazio globalari lotua dagoela (eta ez Europan egindako ebaluazioari, besteak bezalaxe). Y* ikurrak esan nahi du ez dagoela TAC zehatzik espezie horretarako. Espezie horiek TAC kudeaketa unitate orokor bati loturik daude. Horren barnean daude ia arraia espezie gehienak.

	Azken sailkapena																	
		Mozkina			Harrapaketak			Arriskua			Datuak					Informazioa		
Espeziea		**R**	€	%	**R**	Kg	%	**R**	P	M	€/Kg	L (Kg)	L (%)	B (kg)	B (%)	I	IU	C
Legatza	1	1	3E+8	50,73	1	8E+7	36,21	9	3,80	A	3,33	8E+7	53,26	5E+6	6,24	1	LC	Y
Zapo beltza	2	6	1E+7	2,87	8	4E+6	1,65	6	3,91	A	3,95	4E+6	2,55	6E+4	0,08	1	LC	Y
Haitzetako barbarina	3	2	4E+7	6,98	7	4E+6	1,80	22	3,35	A	8,67	4E+6	2,82	4E+3	0,00	5	DD	N
Katuarraina	4	18	5E+6	0,87	4	1E+7	5,48	6	3,91	A	0,94	5E+6	3,24	8E+6	9,34	3	LC	N
Txipiroia	5	3	4E+7	6,82	10	4E+6	1,52	16	3,52	A	10,02	4E+6	2,38	4E+3	0,00	N	DD*	N
Zapo zuria	6	5	2E+7	3,42	11	3E+6	1,32	9	3,80	A	5,9	3E+6	2,03	7E+4	0,08	1	LC	Y
Txitxarro beltza	7	21	3E+6	0,63	2	4E+7	15,10	15	3,61	A	0,83	4E+6	2,64	3E+7	36,73	1	LC	Y
Berdela	8	9	1E+7	2,14	3	3E+7	12,88	29	3,17	E	1,05	1E+7	7,15	2E+7	22,79	1	LC	Y
Oilarra	9	4	2E+7	3,73	16	2E+6	0,97	23	3,34	A	8,85	2E+6	1,47	7E+4	0,08	1	LC	Y
Paneka handia	10	10	1E+7	1,88	6	6E+6	2,40	25	3,28	A	1,82	5E+6	3,60	3E+5	0,29	N	LC	N
Lupia arrunta	11	8	1E+7	2,61	25	1E+6	0,51	9	3,80	A	11,4	1E+6	0,80	0,00	0,00	1	LC	N
Arraia gastaka	11	20	3E+6	0,64	18	2E+6	0,84	5	4,01	A	1,8	2E+6	1,24	1E+5	0,14	3	NT	Y
Gelba	12	40	3E+5	0,05	50	1E+5	0,05	1	4,24	A	2,14	1E+5	0,08	0,00	0,00	5	VU	Y*
Itsas aingira	13	17	5E+6	0,91	15	3E+6	1,10	9	3,80	A	1,89	2E+6	1,69	5E+4	0,06	N	LC	N
Muxumartina	14	13	7E+6	1,36	26	1E+6	0,49	9	3,80	A	6,32	1E+6	0,75	2E+4	0,03	N	DD	N
Txibia	15	7	1E+7	2,66	14	3E+6	1,19	32	3,00	E	4,99	3E+6	1,86	1E+4	0,02	N	LC*	N
Abixoia	16	11	9E+6	1,69	13	3E+6	1,21	26	3,27	A	4,48	2E+6	1,32	9E+5	1,02	N	LC	N
Toil lisoa	17	28	2E+6	0,30	37	4E+5	0,16	4	4,11	A	4,12	4E+5	0,25	2E+3	0,00	3	VU	Y*

Aurreko orrialdearen jarraipena

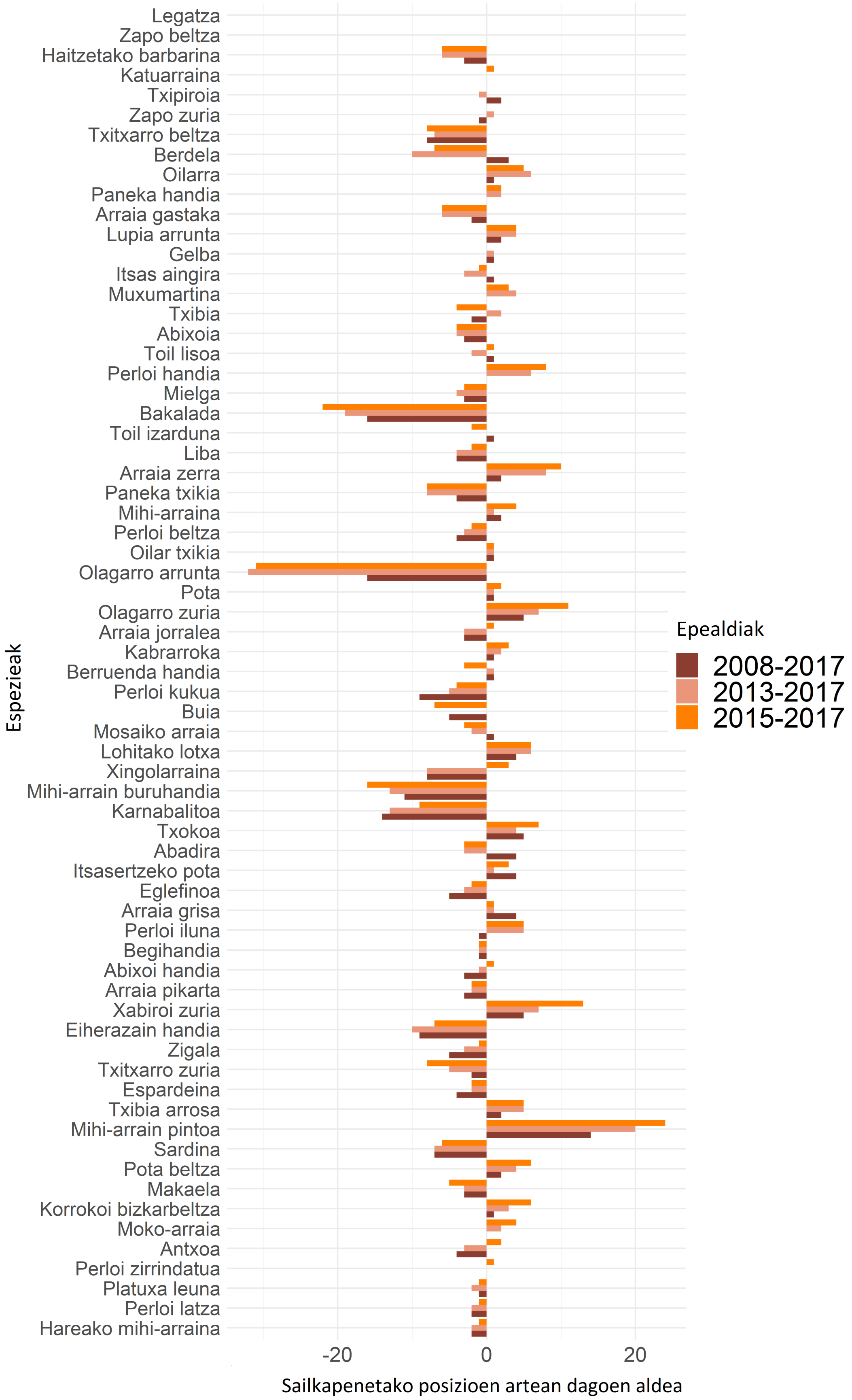
Taula 2.6 – Aurreko orrialdearen jarraipena

Espeziea		Azken sailkapena: Mozkina R	Mozkina €	Mozkina %	Harrapaketak R	Harrapaketak Kg	Harrapaketak %	Arriskua R	Arriskua P	Arriskua M	Datuak €/Kg	Datuak L (Kg)	Datuak L (%)	Datuak B (kg)	Datuak B (%)	Informazioa I	Informazioa IU	Informazioa C
Perloi handia	18	12	7E+6	1,44	19	2E+6	0,73	19	3,43	A	4,44	2E+6	1,13	2E+4	0,03	N	LC	N
Mielga	19	47	7E+4	0,01	54	6E+4	0,03	2	4,23	A	1,57	4E+4	0,03	2E+4	0,03	1	EN	Y
Bakalada	20	33	8E+5	0,16	5	8E+6	3,31	31	3,10	E	1,36	6E+5	0,40	7E+6	8,36	1	LC	Y
Toil izarduna	21	39	3E+5	0,06	48	2E+5	0,08	3	4,13	A	2,47	1E+5	0,09	4E+4	0,05	3	NT	Y*
Liba	22	22	3E+6	0,49	17	2E+6	0,93	16	3,52	A	1,33	2E+6	1,28	3E+5	0,32	5	LC	Y
Arraia zerra	23	29	1E+6	0,21	22	1E+6	0,53	10	3,76	A	1,43	8E+5	0,51	5E+5	0,58	3	LC	Y
Paneka txikia	24	19	4E+6	0,76	12	3E+6	1,21	37	2,83	E	2,03	2E+6	1,31	9E+5	1,04	N	LC	N
Mihi-arraina	25	15	2E+6	1,02	39	4E+5	0,16	15	3,61	A	15,76	3E+5	0,23	3E+4	0,03	1	LC	Y
Perloi beltza	25	25	5E+6	0,42	9	4E+6	1,62	39	2,76	E	2,04	1E+6	0,71	3E+6	3,18	N	LC	N
Oilar txikia	26	14	6E+6	1,18	30	8E+5	0,34	27	3,25	A	7,97	8E+5	0,52	2E+4	0,03	5	LC	Y
Olagarro arrunta	27	16	5E+6	0,97	21	1E+6	0,54	36	2,87	E	4,46	1E+6	0,76	1E+5	0,14	N	LC*	N
Pota	28	26	2E+6	0,38	24	1E+6	0,51	22	3,35	A	2,09	9E+5	0,63	3E+5	0,31	N	LC	N
Olagarro zuria	29	23	2E+6	0,48	23	1E+6	0,53	28	3,25	A	2,08	1E+6	0,81	4E+4	0,04	N	LC*	N
Arraia jorralea	30	57	2E+3	0,00	64	1E+3	0,00	5	4,01	A	2,02	1E+3	0,00	0,00	0,00	6	VU	Y*
Kabrarroka	31	42	2E+5	0,04	53	7E+4	0,03	9	3,80	A	3,68	6E+4	0,04	1E+4	0,01	N	LC	N
Berruenda handia	32	37	5E+5	0,10	42	3E+5	0,11	13	3,67	A	1,95	3E+5	0,18	8E+2	0,00	3	LC	Y
Perloi kukua	33	49	4E+4	0,01	28	1E+6	0,41	16	3,52	A	2,17	2E+4	0,01	9E+5	1,10	6	LC	N
Buia	34	43	2E+5	0,04	32	6E+5	0,26	16	3,52	A	2,29	8E+4	0,05	5E+5	0,62	N	NE	N
Mosaiko arraia	35	54	7E+3	0,00	61	1E+4	0,00	7	3,89	A	2,02	3E+3	0,00	7E+3	0,01	6	NT	Y
Lohitako lotxa	36	36	6E+5	0,11	44	2E+5	0,10	15	3,61	A	3,2	2E+5	0,12	4E+4	0,04	3	DD	N
Xingolarraina	37	38	5E+5	0,09	43	3E+5	0,11	17	3,50	A	2,06	2E+5	0,16	3E+4	0,03	N	LC	N
Mihi-arrain buruhandia	38	24	2E+6	0,48	36	4E+5	0,18	33	2,96	E	7,33	3E+5	0,23	8E+4	0,09	N	LC	N
Karnabalitoa	39	60	0,00	0,00	29	9E+5	0,40	17	3,50	A	0,85	0,00	0,00	9E+5	1,09	3	LC	N
Txokoa	40	27	2E+6	0,31	46	2E+5	0,08	24	3,30	A	8,6	2E+5	0,13	5E+3	0,01	N	DD*	N
Abadira	41	34	6E+5	0,12	49	1E+5	0,06	18	3,46	A	4,66	1E+5	0,09	2E+2	0,00	5	LC	Y
Itsasertzeko pota	42	35	6E+5	0,12	35	4E+5	0,18	25	3,28	A	2,07	3E+5	0,20	1E+5	0,14	N	LC*	N

Aurreko orrialdearen jarraipena

Taula 2.6 – Aurreko orrialdearen jarraipena

		Azken sailkapena																
		Mozkina			Harrapaketak			Arriskua			Datuak					Informazioa		
Espeziea		**R**	€	%	**R**	Kg	%	**R**	P	M	€/Kg	L (Kg)	L (%)	B (kg)	B (%)	I	IU	C
Eglefinoa	43	55	5E+3	0,00	52	1E+5	0,04	11	3,70	A	1,4	4E+3	0,00	1E+5	0,12	N	LC	N
Arraia grisa	44	59	1E+1	0,00	67	9E+0	0,00	8	3,86	A	1,15	9E+0	0,00	0,00	0,00	6	CR	Y*
Perloi iluna	45	41	2E+5	0,04	33	5E+5	0,22	26	3,27	A	1,22	2E+5	0,13	3E+5	0,37	N	LC	N
Begihandia	46	45	1E+5	0,03	57	3E+4	0,01	15	3,61	A	5,65	2E+4	0,02	2E+3	0,00	N	LC*	N
Abixoi handia	47	60	0,00	0,00	56	4E+4	0,02	12	3,69	A	4,57	0,00	0,00	4E+4	0,05	3	LC	N
Arraia pikarta	48	51	2E+4	0,00	58	2E+4	0,01	14	3,65	A	1,96	1E+4	0,01	9E+3	0,01	3	LC	Y*
Xabiroi zuria	49	32	9E+5	0,17	31	8E+5	0,33	42	2,57	B	1,86	5E+5	0,32	3E+5	0,36	N	LC	N
Eiherazain handia	50	60	0,00	0,00	20	2E+6	0,66	37	2,83	E	0,61	0,00	0,00	2E+6	1,81	N	LC	N
Zigala	51	46	1E+5	0,02	59	1E+4	0,01	19	3,43	A	17,49	7E+3	0,00	5E+3	0,01	1	LC*	Y
Txitxarro zuria	52	44	2E+5	0,03	41	3E+5	0,13	29	3,17	E	0,82	2E+5	0,14	1E+5	0,13	N	LC	Y
Espardeina	53	60	0,00	0,00	45	2E+5	0,09	20	3,40	A	55,47	0,00	0,00	2E+5	0,25	N	LC*	N
Txibia arrosa	54	31	1E+6	0,18	51	1E+5	0,05	35	2,91	E	9,35	1E+5	0,07	1E+4	0,01	N	DD*	N
Mihi-arrain pintoa	55	30	1E+6	0,20	47	2E+5	0,08	41	2,64	E	6,29	2E+5	0,11	3E+4	0,03	N	LC	N
Sardina	56	60	0,00	0,00	27	1E+6	0,47	37	2,83	E	1,25	0,00	0,00	1E+6	1,28	2	NT	N
Pota beltza	57	50	3E+4	0,01	60	1E+4	0,01	22	3,35	A	2,68	1E+4	0,01	1E+3	0,00	N	LC*	N
Makaela	58	53	1E+4	0,00	34	5E+5	0,19	37	2,83	E	0,6	2E+4	0,02	4E+5	0,50	N	LC	N
Korrokoi bizkarbeltza	59	48	6E+4	0,01	38	4E+5	0,16	38	2,80	E	1,21	5E+4	0,03	3E+5	0,39	N	LC*	N
Moko-arraia	60	58	8E+1	0,00	66	4E+1	0,00	21	3,39	A	1,83	4E+1	0,00	0,00	0,00	6	NT	Y*
Antxoa	61	60	0,00	0,00	40	4E+5	0,16	40	2,69	E	2,05	0,00	0,00	4E+5	0,43	1	LC	Y
Perloi zirrindatua	62	52	2E+4	0,00	55	6E+4	0,03	35	2,91	E	1,55	1E+4	0,01	5E+4	0,06	N	DD	N
Platuxa leuna	63	60	0,00	0,00	62	1E+4	0,00	30	3,14	E	6,71	0,00	0,00	1E+4	0,01	5	LC*	Y
Perloi latza	64	60	0,00	0,00	63	1E+3	0,00	34	2,94	E	2,55	0,00	0,00	1E+3	0,00	N	LC	N
Hareako mihi-arraina	65	56	3E+3	0,00	65	4E+2	0,00	43	2,55	B	7,31	4E+2	0,00	0,00	0,00	N	LC	N



Irudia 2.5: Aztertutako epealdi ezberdinek sailkapenean duten eragina. Lehorreratzeetarako eta bazterkinetarako azterturiko erreferentziazko epealdia: 2003–2017.

2.4 Eztabaida

Agintariek zientzia-aholku argiak behar dituzte ikerketa eta kudeaketa esfortzuak beharrik handiena duten espezieetan zentratzeko (Smith et al., 2009). Kapitulu honetan, erronka horri erantzuten dion metodo bat proposatu dugu, arrantzaren eragina pairatzeko arrisku potentzial handia duten, ekonomikoki oso garrantzitsuak diren eta/edo arrantza-flotak asko ustiatzen dituen espezieak identifikatzeko. Kapitulu honetan proposatzen den espezieak sailkatzeko metodoa Osio et al. (2015)k proposatutako metodoan oinarritzen da, Osio et al. (2015)k ez bezala bazterkinak ere barnean hartuz. Metodo hau, euskal arrantza-flota demertsal mistoari aplikatu zitzaion. Honela, euskal arrantza-flota demertsal mistoak ustiatzen dituen espezieak sailkatu genituen, ondokoan oinarrituta: PSAk ematen duen espezie bakoitzaren arrisku potentziala, beren ekarpena flotaren mozkinei eta beren harrapaketen bolumena. Arrisku potentzialak euskal arrantza-flota demertsal misto osoa hartzen du aintzat, Micheli et al. (2014)ren proposamena jarraituz, flota ezberdinen suszeptibilitate metatua kontuan hartzen delarik.

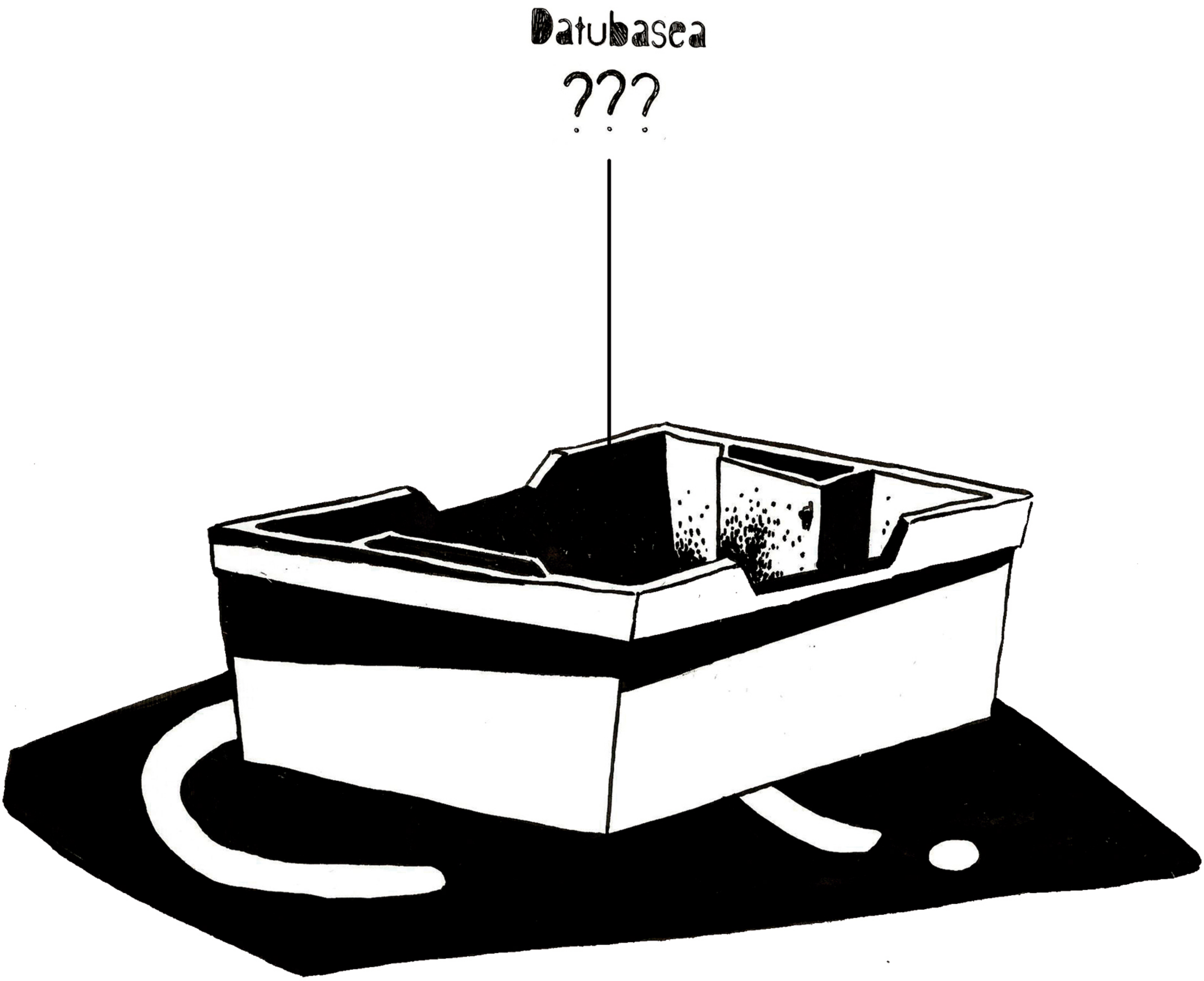
PSAren erabilera oso hedatua badago ere, ez dago gomendio adostu bat bere aplikaziorako. Metodoa kasu bakoitzera egokitzen da eta kasuaren arabehera metodologia ezberdina erabiltzen da arrisku potentziala kalkulatzeko (Hordyk and Carruthers, 2018). Gure azterlana Hobday et al. (2011)k proposatutako PSA atributuetan oinarritu zen. Berak proposatutako PSAn, espezieen arrisku potentziala kalkulatzeko zazpi produktibitate-atributu eta lau suszeptibilitate-atributu puntuatzen dira. Maila trofikoaren atributua kenduta, Hobday et al. (2011)k proposatu zituen beste sei produktibitate-atributuak gure analisian sartu genituen. Maila trofikoaren atributua baztertu genuen, ez zelako espezie guztien produktibitatea neurtzeko erabilgarria. Helduaro-adina, luzera maximoa eta helduaro-luzera atributuen artean korrelazio handia zegoen arren, erredundantzia-analisi bakoitzean kalkulatu ziren aldeak (produktibitate orokorrari eta datuen kalitateari dagokienez) oso txikiak ziren. Honela, ez genuen ebidentziarik aurkitu korrelazionaturik zeuden atributu horietako bat kentzeko eta/edo aukeratzeko, eta guztiak erabili genituen gure PSAn. Ikertzaile batzuk baieztatu dute atributu gehiago sartuz gero produktibitatea modu osatuagoan azter daitekeela (Patrick et al., 2009). Badaude gure lanean erabilitakoak baino atributu gehiago erabiltzen dituzten PSA metodoak (Patrick et al., 2010), baina atributu horiek korrelazioan daude sarritan eta neurtzen zailak dira (heriotza-tasa, adibidez) (Duffy and Griffiths, 2017). Honela, pentsa daiteke korrelazioan dauden atributu horien datu kalitate-maila baxuagoa dela eta, ondorioz, erredundanteak izango direla. Beste zenbait ikertzailek emankortasuna eta ugaltzeari loturiko beste atributu batzuk ez dituzte kontuan hartzen PSA aplikatzeko garaian (Hordyk and Carruthers, 2018), korrelazio faltagatik emankortasunaren, bizitzako lehen urteetan bizirauteko gaitasunaren, eta errekrutamenduaren artean. Gure PSAn, espezieen ugaltze-gaitasun potentziala hartu genuen kontuan, espezieen produktibitatea errundako arrautzen kopuru gisa; baina ez errekrutatzea, hau bai, beste faktore batzuen menpe dagoelako (ingurumen-baldintzak, harrapakariak, etab.). Produktibitate-atributuen atalaseak zehazteko metodoa, ikertzailearen arabera aldatzen da. Kapitulu honetan, erabilienen artean dauden bi metodo aztertu genituen, zentil-metodoa eta K-mean metodoa. Kasu honetan, espezieak oso ezberdinak dira, zefalopodoetatik hasi eta marrazoetaraino. Beren balioak, ondorioz, oso ezberdinak ziren atributu bakoitzaren barruan, nahiz eta espezie gehienak balio-tarte zehatz baten inguruan egon. Honela, zentil-metodoa K mean metodoa baino erabilgarriagoa izan zen espezieen artean dagoen produktibitatea bereizteko. Izan ere, K-mean metodoak, muturreko balioak zituzten espezieak soilik nabarmendu zituen, espezie gehienak produktibitate-kategoria berberean sartuz. Puntuazio-sistemaren alternatiba gisa, zenbaitetan, matematika-ekuazioak erabiltzen dira produktibitate- eta suszeptibilitate-puntuazioak kalkulatzeko (Kirby, 2006). Metodo horien aplikazioa, baina, mugatua da ezezagunak diren datuetarako. Kirby (2006)k proposatu zuen PSA metodoa, esaterako, aplikaezina da harrapaketen datuen batez besteko luzerarik gabe. Honela, ezin dugu aztertutako espezie guztien suszeptibilitatea kalkulatu Kirby (2006)k proposatu zuen PSA erabiliz, datu horiek ezezagunak direlako espezie gehienen kasuan. Datuak faltan daudenean, atributuei ematen zaien trataera ere PSA metodoaren arabera aldatzen da (Hobday et al., 2011; ICES, 2013; Patrick et al., 2010; Smith et al., 2009). Ikertzaile batzuk prebentzio gisa, puntuaziorik altuena esleitzen diote falta den atributuari (Hobday et al., 2011; ICES, 2013). Baina metodo horrek kritika asko jaso ditu, arriskuaren balioak gehiegi balioesteagatik (positibo faltsua) (Hordyk and Carruthers, 2018; Zhou et al., 2016). Gure analisian, faltan zeuden atributuak datuen kalitate-indizea aplikatuz landu genituen (Lucena-Fredou et al., 2017; Patrick et al., 2009). Ez genuen emankortasunari buruzko daturik espezie batzuen kasuan edo tretzaontzien selektibitateari buruzko daturik espezie guztien kasuan. Horiek kenduta, datu kalitate-maila espezie guztietarako nahikoa edo ona zen.

PSAren emaitzak beste metodo batzuen emaitzekin alderatu badira ere (ICES, 2013; Lucena-Fredou et al., 2017; Zhou et al., 2016), lehen aldiz, Hordyk and Carruthers (2018)k ebaluatu zuen PSA metodoa kuantitatiboki. Hordyk and Carruthers (2018)k ondorioztatu zuten espezieen egoera-baldintzarik onenetan, PSAk arriskuaren kategorizazioan izan zezakeen arrakasta-probabilitatea %66koa zela; eta %50etik beherakoa baldintza zehatz batzuetan (esaterako, ustiatze altua eta hasierako populazioaren tamaina txikiegia zenean). Baieztatu zuten, arrisku-ebaluazio kuantitatibo bat garatzeko beharrezkoa den informazioa, PSA bat egiteko beharrezkoa den horren antzekoa dela. Honela, Hordyk and Carruthers (2018)k arriskua ebaluatzeko eredu kuantitatiboak erabiltzea gomendatzen du: simulazio-ereduak, adibidez. Baina, praktikan, arriskua metodo kuantitatiboa erabiliz ebaluatzea ataza zaila eta neketsua izan daiteke datu-mugatuak dituzten espezie ezberdin ugari daudenean (Bentley, 2014). Halaber, datu-mugatuak dituzten espezieek balio txikia dute gehienetan eta, horrenbestez, espezie horiek ebaluatzeko eskura izaten dituzten bitartekoak eta denbora ere mugatuak dira (Bentley, 2014). Eta are gehiago, PSA metodoan erabilitako datuak ez dira nahikoak, askotan, Hordyk and Carruthers (2018)k aplikatu zuen simulazio-ariketa egiteko. PSA metodoek beren balioa egiaztatu dute ariketa berberaren barruan datu-mugatuak dituzten espezieen eta datu-ugari dituzten espezieen arrisku potentziala zehazteko (Ormseth and Spencer, 2011). Halaber, erreminta erabilgarritzat jo da arrisku altua duten espezieak identifikatzeko, eskuragarri dauden datuak mugatuak direnean gehienbat, ICES (2012) eta MSC (2001) erakundeen esanetan. Izan ere, ebaluazio indibidualen kasuan PSAk emandako emaitzak tentu handiz interpretatu beharko balirateke ere, espezieen arteko arrisku-alderaketa erlatiboak sendoak dira (Micheli et al., 2014). Kudeaketa planen ebaluazioan, PSA, ebaluazio-eredu erdi-kuantitatibo erabilgarria da, simulazio-ereduetan sartu beharko liratekeen espezieak zehazteko.

Kapitulu honetan, espezieak lehenesteko metodoa proposatzen da. Honek espezieak sailkatzea ahalbidetzen du, gehien ustiatzen direnetatik, egoera zaurgarrienean daudenetatik eta/edo ekonomikoki garrantzitsuenak direnetatik beste muturrean daudenetaraino. Sailkapenean erabilitako aldagaiek azken sailkapenari egiten dioten ekarpena orekatua zegoen eta aztertutako epealdiak ez zuten eraginik metodoaren emaitzan. Harrapaketen aldagaia arrantza-jardueraren presioa neurtzeko erabili genuen, onarturik harrapatutako espezie guztiak hiltzen direla. Baina harrapatuak izan ondoren marrazoek eta arraiek bizirauteko duten gaitasun handia aintzat hartu beharko litzateke. Beste aukera bat da proxy aldagai bat sartzea, espezieen oparotasun erlatiboak sartuz harrapaketen ordez (Arrizabalaga et al., 2011). Baina aztertutako espezie gehienen kasuan eskura genituen datuak mugatuak ziren eta ez genuen beren oparotasuna ezagutzen.

Arrantza-flota mistoak harrapatzen dituen espezieak asko direla ikusita, praktikan ia ezinezkoa da espezie horiek guztiak simulazio-ereduan sartzea. Ondorioz, simulazioan sartu beharreko espezieak zehazteko espezieen bizi-historia eta arrantza-flotaren jarduera aintzat hartzen dituzten metodoak —hemen aurkeztu duguna esaterako— iragazte-erreminta erabilgarriak dira. Sailkapeneko lehen postuetan dauden espezieak sartzea gomendatzen dugu, harrapaketa guztien eta flotaren mozkinen proportzio handi bat ordezkatzen dutela eta beste espezieei loturiko arriskua ez dela arrantza-flotaren jardueraren eraginpean eroriko ziurtatuz. Kasu honetan, sailkapeneko lehen hamar espezieak simulazio-ereduan sartuz, euskal arrantza-flota demertsal mistoaren kudeaketa planen ebaluazio egokia bermatzen da. Sailkapeneko lehen hamar espezieek euskal arrantza-flota demertsalaren mozkinen, lehorreratzeen eta bazterkinen %76 suposatzen diote. Espezie hauek guztiak (berdela kenduta) arrisku potentzial altua erakutsi zuten. Gehienbat, esfortzu handiagoa egin beharko litzateke haitzetako barbarinaren, katuarrainaren eta txipiroiaren kudeaketa aproposean aurrera egiteko. Hirugarren, laugarren eta bosgarren posizioan daude, hurrenez hurren, sailkapen orokorrean. Halere, ez dago inolako harrapaketa-mugarik edo kuotarik beren ustiaketa erregulatzeko, datu-mugatuak dituzten espezieak sailkatzen diren ICES kategoria batean daude sailkatua eta arrisku potentzial handia agertzen dute. Horrez gain, haitzetako barbarinaren harrapaketek pisu erlatiboa galdu dute, azken urteotan. Arrain guztiak portura eramateko betebehar berriaren eskutik, arreta handiagoa eskaini beharko diegu marrazoei eta arraiei, itoguneko espezie bilakatzeko potentziala baitute beren TAC baxuaren ondorioz.

Espezieak lehenesteko metodoa, euskal flota demertsal mistoei soilik aplikatu zitzaien, Bizkaiko Golkoan jarduten zuten euskal flota demertsal mistoen harrapaketei buruzko datu historiko zehatzak genituelako (espezie eta flota mailan) baina ez, ordea, Espainiako eta Frantziako flota demertsal osorako. Bizkaiko Golkoaren Iparraldean jarduten duen (ICES 8abd) Espainiako flota demertsal mistoaren gehiena euskalduna da eta, horrenbestez, analisian Espainiako flota ia osoa kontuan hartu zen. Kapitulu honetan lortutako espezieen rankinga ICESen arrantza-metodoen lantaldean aurkeztu zen 2018an, eta sailkapena Bizkaiko Golkoan jarduten zuten arrantza-flota demertsal misto guztirako egokia zela ondorioztatu zen, Frantziako eta Espainiako floten dirusarrerei egiten zieten ekarpenari zegokienez garrantzitsuenak ziren espezie gehienak sailkapen horretako lehen postuetan baitzeuden.



3 Kapitulua

Datuen garrantzia SRA ereduan

The English version of this chapter is in Appendix D

3.1 Sarrera

Spezieen populazioak adinean oinarrituta ebaluatu ahal izateko ereduak XX.mende erdialdean garatu ziren lehen aldiz (Hilborn and Walters, 1992). Populazioaren Analisi Birtuala (Virtual Population Analysis, VPA) (Gulland, 1965; Pope, 1972), arrantzafloten kudeaketarako erabili zen adinean oinarritutako lehen ebaluazio-eredua da (Dichmont et al., 2016a). Gaur egun erabiltzen da oraindik. VPA ereduaren aldaera ezberdinak daude. Beren arteko ezberdintasuna, datuak erabiltzeko moduarekin eta emaitza numerikoa lortzeko erabiltzen duten algoritmoarekin erlazionatuta dago (esaterako, findu gabeko VPA, ad hoc findutako VPA eta estatistikoki egokitutako VPA). VPA aldaera guztietarako, adinaren araberako harrapaketen datuak behar dira eta datu hauek ez daude eskuragarri ustiatutako populazio askorentzat. VPA ereduaren alternatiba gisa, adinaren araberako harrapaketen daturik ez dagoen kasuetarako, SRA eredua garatu zuten Kimura and Tagart (1982)k. Geroago, Walters et al. (2006)ek SRA eredua osatu zuten, Monte Carlo simulazioak SRA ereduan sartuz. Honek, populazio-dinamiken probabilitate-banaketa sortzeko aukera ahalbidetu zion SRA ereduari, populazio atributuei (adibidez, heriotza-tasa naturala edo arrantza-jardueraren selektibitatea) loturiko hipotesi ezberdinak oinarri hartuta. Kimura and Tagart (1982)k eta Walters et al. (2006)ek SRA eredua aplikatzeko ezarri zuten hipotesia da, populazioa arrantzarik gabeko egoeran dagoela, eredua aplikatzen den lehen urtean. Baina baliteke, harrapaketen datuak eskuragarri ez egotea arrain-populazio baten ustiatzearen hasiera-hasieratik. Honen aurrean, Huynh et al. (2020a)ek SRA ereduaren aplikazio berri bat garatu zuen. Honi esker, SRA eredua ustiatze-egoera batetik abia daiteke. Ustiatze-egoera batean, ereduaren lehen urteko populazioa kalkulatzeko, populazioa, SRA ereduaren erabiltzaileak zehazten duen harrapaketa-maila batekin orekan dagoela onartzen da; hau da, kalkulatzen den ereduaren lehen urteko populazioa, erabiltzaileak zehazten duen harrapaketa-maila hori denboran zehar mantentzeko gai izango da bere biomasa-maila aldatu gabe (aurrerantzean, “harrapaketa-biomasa oreka”). Harrapaketa-biomasa orekari buruzko hipotesia hasierako populazio-proxy gisa, populazioak ebaluatzeko beste eredu batzuetan ere erabiltzen dira: oso hedatua dagoen Stock Sintesi ebaluazio-ereduan (Methot and Wetzel, 2013), adibidez.

Populazioa ebaluatzeko eredurik egokiena, eskuragarri dauden datuen menpe dago, beste alderdi batzuren artean (Dichmont et al., 2021). Guztizko harrapaketen eta abundantzia-indizearen datuek, populazioaren tamainaren eta joeraren inguruko informazioa eskaintzen dute, hurrenez hurren; luzeraren eta/edo adinaren araberako datuek, beren aldetik, populazioaren tamainari buruzko informazioa eskaintzeaz gain, ondoko alderdiak hobeto ulertzen laguntzen dute: populazio-egitura, heriotza-tasa naturala, hazkundea eta errekrutatzea (Chen et al., 2003; Magnusson and Hilborn, 2007). Ikerketa ezberdinek frogatu dutenez, ebaluazio-ereduaren emaitzen zehaztasuna faktore hauen menpe dago: erabilitako datuen oinarria (hau da, biomasan agregatuak edo adinaren edo luzeraren araberakoak) (Chen et al., 2003; Magnusson and Hilborn, 2007; Ono et al., 2014; Wetzel and Punt, 2011), datuak bildu diren epealdiaren luzera (Chen et al., 2003; Ono et al., 2014; Wetzel and Punt, 2011), datuen banatzea eta lagin-tamaina (Fisch and Bence, 2020; He et al., 2016; Hulson et al., 2017; Muradian et al., 2019), harrapaketen heriotza-tasaren joera populazioan (Magnusson and Hilborn, 2007; Ono et al., 2014) eta espeziearen biologia-parametroak (Ono et al., 2014). Ikerketa horietako zenbaitek erabili zuten hipotesia izan zen populazioa ustiatze-egoeran zegoela datuak hasten ziren lehen urtean (Fisch and Bence, 2020; He et al., 2016; Hulson et al., 2017; Muradian et al., 2019). Baina ikerketa horietan aztertutako agertokietan, luzeraren edo adinaren araberako informazioa sartu zen, ebaluazio-eredu konbentzionalen erabilera ahalbidetuz hasierako populazioa kalkulatzeko. Beste ikerketek erabili zuten hipotesia izan zen, populazioa arrantzarik gabeko egoeran zegoela ebaluazio-ereduaren lehen urtean (Chen et al., 2003; Ono et al., 2014; Wetzel and Punt, 2011). Hasierako populazioari buruzko hipotesiak ereduaren aplikazioan duen eragina ulertzea erronka garrantzitsua da oraindik.

"Auto-proba" simulazioa, ebaluazio-ereduen errendimendua ezagutzeko funtsezko analisia da, parametroak egokiro kalkulatzeko duen gaitasunari dagokionez (Deroba et al., 2014; Punt et al., 2020). Hordyk et al. (2019)ek SRA ereduaren errendimendua aztertu zuen biologia-parametroetan (hau da, heriotza-tasa naturalean, helduaro-adinean eta errekrutatze makurduran) alborapena eta behaketa- eta eredu-hipotesietan akatsak (hau da, abundantzia-indize orekatuegia edo lermatuegia, gehiegizko edo gutxiegizko harrapaketak, eta selektibitate-kurbaren itxura okerra) sartuz. Baina orain arte ez da kuantitatiboki ebaluatu epealdi-luzera ezberdinetako datuak, datu-egitura ezberdinak eta hasierako populazioari loturiko hipotesi alternatiboak erabiliz SRA ereduak agertzen duen errendimendua.

Kapitulu honen xedea, "auto-proba" simulazioaren bitartez SRA ereduaren errendimendua ebaluatzea da harrapaketen eta abundantzia-indizearen epealdi-luzera eta datu-egitura, eta hasierako populazioaren inguruko hipotesi ezberdiak erabiliz. Gainera, emaitzek populazioaren ustiatze-maila ezberdinekiko eta arrantza-jarduerarén selektibitateari buruzko hipotesi ezberdinekiko agertzen duten sentikortasuna ere aztertu da. Emaitza hauei esker, modu fidagarrian SRA eredua zein kasutan erabil daitekeen, datu-mugatuak dituzten espezieen dinamika simulatzeko, identifikatu da.

3.2 Materialak eta metodoak

SRA ereduaren errendimendua "auto-proba" simulazioz ebaluatzeko, "egiazko" populazio bat sortu zen, behaketa-akatsa gehitu zitzaion "egiazko" populazio horri, eta SRA eredua hausaz sorturiko datu-multzoetara egokitu zen. Datu-multzo horiek sortzeko, datuen eskuragarritasunari buruzko agertoki ezberdinak, hasierako populazioari loturiko hipotesi ezberdinak, eta arrantza-jarduerarén selektibitateari loturiko datu zehaztugabe ezberdinak aintzat hartu ziren. "Egiazko" eta kalkulatutako parametroen arteko koherentzia aztertzeko, zehaztasun eta alborapen errendimendu-estatistikak erabili ziren.

3.2.1 Simulazio-esparrua

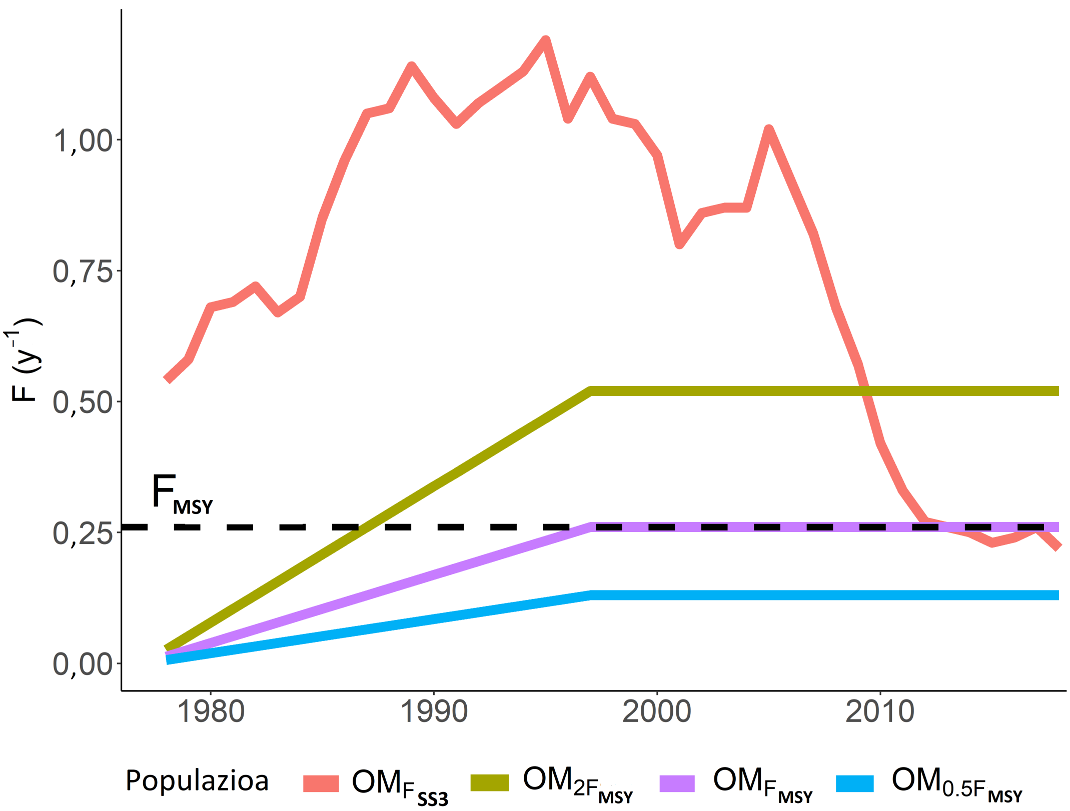
Simulazio-algoritmoak hiru osagai zituen: eredu eragilea (OM), behaketa-eredua eta ebaluazio-eredua. OMak "egiazko" populazioa eta ustiatze-dinamikak irudikatzen zituen bitartean, behaketa-ereduak OMaren harrapaketen eta abundantzia-indizearen inguruko hipotesiak sortu zituen. Ebaluazio-ereduan, SRA eredua aplikatu zitzaien behaketa-ereduak sorturiko datuei. Zehazki, OM bakoitzerako 100 populazio erreplika sortu ziren, errekrutatze-desbideratze ezberdinekin. Ondoren, behaketa-datuen multzoa sortu zen populazio-erreplika bakoitzerako, behaketa-akatsarekin. Azkenik, SRA eredua aplikatu zitzaion datu-multzo bakoitzari, 100 kalkulu sortuz honela.

Simulazioa, 4., 6. eta 7. ICES azpiguneetan eta 3a eta 8abd ICES dibisioetan aurki daiteen legatzaren (*Merluccius merluccius*) populazioaren ebaluazioan oinarritu zen (ICES, 2019a). Espezie hau, demertsala da eta honen ebaluaketa egiteko, Stock Sintesi izeneko ebaluazio-eredu integratua (Methot and Wetzel, 2013) erabiltzen da.

Ikerketa honetan, OMak eta behaketa-ereduak, Stock Sintesi ebaluazio-ereduaren epealdi-luzera osoa hartu zuten barnean (1978tik 2018ra, 41 urte). Epealdi oso hori, epealdi-luzera motzagoak sortzeko erabili zen agertoki ezberdinetan, datuen luzera ezberdinek SRA ereduaren aplikazioan duten eragina aztertzeko.

3.2.1.1 Eredu eragilea

Adinean oinarritutako lau OM sortu ziren. Lehen populazioari $OM_{F_{SS3}}$ izena eman zitzaion. Populazio hau sortzeko erabilitako harrapaketen heriotza-tasa (1978tik 2018ra) eta hasiera-populazioa (1978koa), legatzaren ebaluazio-eredutik atera ziren (ICES, 2019a). OM honetan, lehen urteko populazioa ustiatze-egoeran zegoen. Beste hiru OMetan, populazioa arrantzarik gabeko egoeratik abiatzen zen. Hiru populazio hauetan, harrapaketen heriotza-tasak ibilbide ezberdina jarraitzen zuen (3.1.Irudia), non honen ondorioz, populazio bakoitzaren murrizte tasa ezberdina zen simulazioaren azken urtean. Hiru populazio hauetan, harrapaketen urteko heriotza-tasa (year^{-1}) linealki hazten zen lehen 20 urtetan eta, ondoren, hurrengo 21 urtetan, aldatu gabe mantentzen zen (3.1.Irudia). Hiru populazioetan, azken urteetako harrapaketen heriotza-tasa, MSY harrapaketen heriotza-tasarekiko (F_{MSY} = 0,26) proportzionala izan zen (ICES, 2019a). F_{MSY}ren hiru biderkatzaileak 2, 1 eta 0,5 izan ziren. Aipatuak $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ eta $OM_{0,5F_{MSY}}$ populazioei dagozkie, hurrenez hurren.



Irudia 3.1: Harrapaketen heriotza-tasa (F) $OM_{F_{SS3}}$, $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ eta $OM_{0,5F_{MSY}}$ populazioetan. Lerro eten horizontal beltzak, legatzaren harrapaketen heriotza-tasa errendimendu maximo jasangarrian (F_{MSY}) irudikatzen du (ICES, 2019a).

$\mathrm{OM}_{2\mathrm{F}_{\mathrm{MSY}}}$, $\mathrm{OM}_{\mathrm{F}_{\mathrm{MSY}}}$ eta $\mathrm{OM}_{0,5\mathrm{F}_{\mathrm{MSY}}}$ populazioen kasuan, ondoko ekuazioa erabili zen hasierako populazioa kalkulatzeko:

$$N_{1,a} = \begin{cases} R_0 & a = 1, \\ R_0\ e^{-M(a-1)} & a = 2 \ldots A-1, \\ \frac{R_0\ e^{-M(A-1)}}{1-e^{-M}} & a = A, \end{cases} \tag{3.1}$$

non $N_{1,a}$ den a adineko banakoen kopurua datuen lehen urtean ($y = 1$, 1978.urteari dagokio), R_0 den arrantzatu gabeko errekrutatzea, M den heriotza-tasa naturala eta Ak plus taldea adierazten duen A adineko banakoak edo zaharragoak diren banakoen kopurua bilduz. R_0, M eta A balioak, legatzaren ebaluazioan erabilitako balio horietan finkatu ziren (ICES, 2019a) (3.1.Taula).

Taula 3.1: Eredu eragilea osatu eta ebaluazioa-eredua aplikatzeko erabili ziren legatzaren bizi-historiari buruzko parametroak. Adin-tartea, von Bertalanffy hazkunde-parametroak (L_∞ luzera asintotikoa, k aldatsa, eta t_0 hasierako tamaina), Beverton eta Holt populazio-errekrutatze erlazio-parametroak (h makurdura, R_0 errekrutatzea arrantzarik gabeko baldintzetan eta σ errekrutatze-desbideratzeen desbideratze estandarra), batez besteko luzeraren eta adinaren arteko bariazio-koefizientea (cv), luzera-pisu erlazioaren parametroak (a eta b), adin-heldutasuna eta heriotza-tasa naturala (M).

Legatza		**Balioa**[*]
Adin tartea		1-15[+]
Hazkunde-parametroak	L_∞ (cm)	130
	k (year^{-1})	0,177319
	t_0 (year)	0
Errekrutatzea	h	0,96
	R_0 (thousands)	273.569,92
	σ	0,4
Luzera (cm) -adina	cv	0,15
Luzera (cm) -pisu (kg)	a	5,13E-6
	b	3,074
Adin-heldutasuna	1/2/3/4/5/>6 age	0,01/0,34/0,80/0,95/0,98/1
Heriotza-tasa naturala	M (year^{-1})	0,4

[*] Balio hauek ICES (2019a)etik hartu ziren. ICES (2019a)en ebaluazioan adin-tartea 0tik 15era arte doa: errekrutatzea 0.adinean gertatzen da eta 15 urteko adina plus taldea da. SRA ereduan errekrutatzea 1.adinean gertatzen denez eta adin-tarteak R_0 eta h ren kalkuluetan eragiten duenez, ICES (2019a)ek erabilitako R_0 (361.272 mila) eta h (0,99) parametroen balioak 1-15 adin tarterako kalkulatu ziren Beverton eta Holt populazio-errekrutatze erlazioaren ekuazioa erabiliz (Beverton and Holt, 1957), datu hauek OM eta ebaluazio-ereduan erabili aurretik.

Lau populazioetan, ondorengo urteetarako adinaren araberako banakoen kopurua kalkulatzeko ($N_{y,a}$), adinean oinarritutako biziraupen esponentzialaren ekuazioa erabili zen (Quinn and Deriso, 1999):

$$N_{y,a} = \begin{cases} R_y & a = 1, \\ N_{y-1,a-1} e^{-Z_{y-1,a-1}} & a = 2,\ \ldots, A-1, \\ N_{y-1,a-1} e^{-Z_{y-1,a-1}} + N_{y-1,a} e^{-Z_{y-1,a}} & a = A, \end{cases} \tag{3.2}$$

non R_y den urteko errekrutatzea eta $Z_{y,a}$ den heriotza-tasa osoa. Heriotza-tasa osoa, heriotza-tasa naturalaren (M) eta adinaren araberako harrapaketen heriotza-tasaren ($F_{y,a}$) batuketa gisa kalkulatu zen. Ekuazio guztietan, y eta a azpiindizeek urteari eta adinari egiten diete erreferentzia, hurrenez hurren.

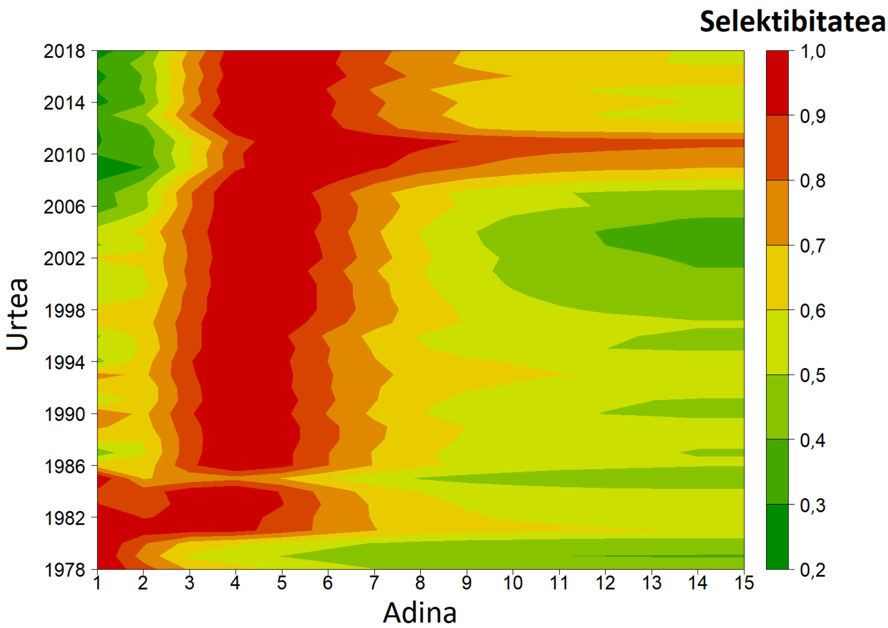
Errekrutatzea (R_y), 1.adinean gertatu zen eta Beverton eta Holt populazio-errekrutatze erlazioaren ekuazioa (Beverton and Holt, 1957) erabiliz kalkulatu zen:

$$R_y = \frac{4 * h * R_0 * SSB_{y-1}}{(SSB_0 * (1-h)) + (SSB_{y-1} * (5 * h - 1))} * exp\left(\delta_y - 0,5 * \sigma^2\right), \delta_y \sim N\left(0; \sigma^2\right), \tag{3.3}$$

non SSB_y den urteko biomasa ugaltzailea, SSB_0 den biomasa ugaltzailea arrantzarik gabeko egoeran, h den makurdura (R_0 ren proportzioa SSB SSB_0 ren 0,2ra murrizten denean), δ_y den urteko errekrutatze-desbideratzea eta σ den errekrutatze-desbideratzeen desbideratze estandarra.

OM bakoitzeko 100 populazio-erreplika sortu ziren δ_y laginak hartuz. δ_y balioek banaketa normala jarraitzen zuten, bere batez bestekoa 0 koa eta desbideratze estandarra σ ren berdina izanda. h, R_0 eta σ balioei, legatzaren ebaluaketan erabiltzen diren balioak esleitu zitzaizkien (3.1.Taula). SSB_0ren balioa, R_0 balioetan, luzera-pisu erlazio-parametroetan (a eta b) eta adin bakoitzean helduek suposatzen duten proportzioetan oinarrituta kalkulatu zen (3.1.Taula).

y urtean a adineko harrapaketen heriotza-tasa ($F_{y,a}$), urteko harrapaketen heriotza-tasaren (3.1.Irudia) eta urteko adinaren araberako arrantza-jardueraren selektibitatearen (3.2.Irudia) arteko produktu gisa kalkulatu zen. Lau populazioak sortzeko ICES (2019a)ek kalkulatu zuen urteko adinaren araberako arrantza-jardueraren selektibitatea erabili zen (3.2.Irudia). Lehen zortzi urteetan (1978tik 1986ra) arrantza-flotak 1.adineko banakoak arrantzatu zituen gehienbat. Azken urteotan (1987tik 2018ra), ordea, 4.adineko banakoak arrantzatu ditu gehienbat (3.2.Irudia).



Irudia 3.2: Legatzaren adinaren araberako arrantza-jardueraren selektibitatea, 1978tik 2018ra arte.

"Egiazko" populazioen adinaren araberako urteko harrapaketak kalkulatzeko, Baranov harrapaketa-ekuazioa (Baranov, 1918) erabili zen. Urteko harrapaketen biomasa guztia kalkulatzeko, adinaren araberako harrapaketen eta adinaren araberako banakoen pisuaren arteko produktuen batuketa egin zen. OMak adinean oinarrituta zeudenez, luzeraren araberako urteko harrapaketak kalkulatzeko, adinaren araberako harrapaketen eta a adinean l luzera izateko probabilitatearen arteko produktua erabili zen. a adinean l luzera izateko probabilitatea $P(l|a)$, honela kalkulatu zen:

$$P(l|a) = \begin{cases} \Phi({L'}_{l+1}), & l = 1, \\ \Phi({L'}_{l+1}) - \Phi({L'}_{l}) & l = 2, \ \ldots, \ L-1, \\ 1 - \Phi({L'}_{l}) & l = L, \end{cases} \tag{3.4}$$

non L'_l den l luzera-edukiontziko beheko mugaren luzera. Luzera-edukiontziak 5,5tik 135,5ra zihoazen 2 cm-ko tarteetan. $\Phi({L'}_l)$ aldagai normal baten banaketa-funtzio metatua da, bere batez bestekoa adinaren araberako luzeraren batez bestekoa da eta desbideratze estandarra adinaren araberako batez besteko luzeraren eta adinaren araberako batez besteko luzeraren bariazio-koefizientearen arteko produktua da.

Adinaren araberako batez besteko luzera kalkulatzeko, 3.1.Taulan jasotako von Bertalanffy hazkunde-ereduaren parametroak eta adinaren araberako batez besteko luzeraren bariazio-koefizientea erabili ziren, ebaluazio-taldeak erabiltzen dituenak horiek baitira (ICES, 2019a). Harrapaketen "egiazko" batez besteko luzera, luzeraren araberako harrapaketa datuen batez besteko haztatuaren berdina da.

3.2.1.2 Behaketa-eredua

"Egiazko" populazioen datuetan behaketa-akatsak sartuta lortu ziren behatutako datuak. Behatutako guztizko harrapaketak eta behatutako biomasa-indizea, "egiazko" harrapaketa guztiei eta biomasa-indizeari akats lognormal bat biderkatuz kalkulatu ziren. Akats lognormal honen batez bestekoa 1 ekoa zen eta bariazio-koefizientea, 0,25ekoa. Adin eta luzera-edukiontzi bakoitzean behatutako banakoen proportzioa, banaketa multinomial batean oinarritu zen; lagin-tamaina eraginkorra (Effective Sample Size, ESS) 50ean ezarri zen adinaren araberako proportzioetarako Magnusson and Hilborn (2007) irizpide hartuz, eta 125ean luzeraren araberako proportzioetarako legatzaren ebaluazioan erabiltzen den bezalaxe (ICES, 2019a). Behatutako luzeraren araberako harrapaketak, adinaren araberako harrapaketak, eta adinaren araberako abundantzia-indizea lortzeko ($\widehat{X_{y,s}}$), ondoko foruma orokorra erabili zen:

$$\widehat{X_{y,s}} \sim \mathit{Multinomial}\ (ESS,\ P(s=1,y),\ \ldots\ ,\ P(s=S,y)) \ * \ \frac{X_y}{ESS}, \tag{3.5}$$

non $\widehat{X_{y,s}}$ den behatutako luzeraren araberako harrapaketen, behatutako adinaren araberako harrapaketen edo behatutako adinaren araberako abundantzia-indizea, s den adina edo luzera-edukiontzia S adin edo luzera-edukiontzi maximoa izanik, $P(s,y)$ den banakoen proportzioa s edukiontzian y urtean, eta X_y den y urteko banakoen "egiazko" kopurua. Proportzioak, luzeraren araberako harrapaketen, adinaren araberako harrapaketen edo adinaren araberako abundantzia-indizearen "egiazko" datuetatik kalkulatu ziren. Harrapaketetan behatutako batez besteko luzera, banaketa lognormal batean oinarrituta kalkulatu zen, zeinaren batez bestekoa "egiazko" luzeraren araberako harrapaketen batez besteko haztatua zen eta bariazio-koefizientea 0,25ekoa.

3.2.1.3 Ebaluazio-eredua

Populazio-abundantzia eta ustiatze maila kalkulatzeko, SRA ebaluazio-eredua erabili zen [MSEtool paketean eskuragarri dagoena -2.0.1 bertsioa- (Huynh et al., 2020a), R ingurunearen barruan (R Core Team, 2022)]. SRA ereduak populazio-dinamika kalkulatzeko erabiltzen dituen ekuazioak, OMn erabilitakoen berdinak dira: biziraupen esponentzialaren ekuazioa, Beverton eta Holt populazio-errekrutatze erlazioaren ekuazioa eta Baranov harrapaketa-ekuazioa (Walters et al., 2006). Halaber, SRA ereduak, OMn erabilitako ekuazio berdinak erabiltzen ditu ondokoak kalkulatzeko: harrapaketen batez besteko luzera, luzeraren araberako harrapaketak, adinaren araberako harrapaketak eta adinaren araberako abundantzia-indizea (Huynh et al., 2020a). SRA ereduan, besterik esan ezean, arrantza-jardueraren selektibitatea konstante mantentzen da denboran zehar. Arrantza-jardueraren selektibitate historikoa ezagutzen ez denean, hau, adinaren edo luzeraren araberako datuetatik kalkulatzen da. Adinaren edo luzeraren araberako daturik erabiltzen ez denean SRA eredua aplikatzeko, adinaren araberako arrantza-jardueraren selektibitatea (sel_a) honela kalkulatzen da:

$$sel_a = \begin{cases} 2^{-[(L_a \; - \; L_{FS})/\sigma_{asc}]^2} & L_a \; < \; L_{FS}, \\ \\ 2^{-[(L_a \; - \; L_{FS})/\sigma_{des}]^2} & L_a \; \geq \; L_{FS}, \end{cases} \tag{3.6}$$

non $\sigma_{asc} = (L_5 - L_{FS})/\sqrt{-log_2\,(0,05)}$ eta $\sigma_{des} = (L_\infty - L_{FS})/\sqrt{-log_2\,(V_{maxlen})}$ selektibitate-kurbaren goranzko eta beheranzko formaren itxura kontrolatzen duten parametroak diren, hurrenez hurren. L_a adinaren araberako batez besteko luzera, L_∞ luzera asintotikoa, L_5 selektibitatearen %5aren luzera, L_{FS} selektibitate osoaren luzera eta V_{maxlen} selektibitatea L_∞ an dira.

SRA ereduaren helburua ondokoa da: biologia-parametroen (luzera-pisu erlazioa, adin-heldutasuna, makurdura, von Bertalanffy hazkundea eta heriotza-tasa naturala), errekrutatze-desbideratzeen desbideratze estandarraren, abundantzia-indizearen selektibitatearen eta arrantza-jardueraren selektibitatearen balioak jakinda, behatutako datu hauek sor ditzazketen hasierako biomasaren, harrapaketen heriotzatasaren, eta errekrutatze-mailaren konbinazio posiblea kalkulatzea. SRA eredua aplikatzeko behar diren gutxieneko datuak, urteko harrapaketa guztiak eta guztizko abundantzia-indizea dira. Hala ere, datu osagarri batzuk ere sar daitezke SRA ereduan: harrapaketen urteko batez besteko luzera, luzeraren araberako harrapaketak, adinaren araberako harrapaketak edo/eta adinaren araberako abundantzia-indizea. SRAk, log-likelihood maximoa erabiltzen du ondoko datuak kalkulatzeko: harrapaketen heriotza-tasa, arrantza-jardueraren selektibitatea, R_0, errekrutatze-desbideratzeak eta harrapaketa-biomasa oreka. Harrapagarritasun-koefizientea (q) analitikoki ebazten da. E.1 Eranskinak SRA ereduak parametroak kalkulatzeko erabiltzen dituen ekuazioak jasotzen ditu. Log-likelihood maximoaren metodoak, harrapaketa guztietarako, abundantzia-indizerako, harrapaketa-biomasa orekari buruzko hipotesirako, errekrutatze-desbideratzerako eta erabilitako beste datu-multzo osagarri guztietarako log-likelihooden batura maximizatzean datza. E.2 Eranskinak log-likelihood osagaiak kalkulatzeko ekuazioak jasotzen ditu. SRA ereduak, datu guztien log-likelihood osagaietan ziurgabetasuna sartzeko aukera ematen du. Salbuespen gisa, harrapaketa guztien bariazio-koefizientea eta harrapaketa-biomasa orekari buruzko hipotesiaren bariazio-koefizientea 0,01 ean finkatzen da, harrapaketa guztietan eta harrapaketa-biomasa orekari buruzko hipotesian oso akats gutxi dagoela onartuz.

SRA eredua aplikatu zenean, ez zen inolako akatsik sartu ez biologia-parametroetan ez eta errekrutatze-desbideratzeen desbideratze estandarrean ere (3.1.Taula). Parametro hauen balioak, OMtik atera ziren. Behatutako biomasa-indizearen datuak, proportzionalak ziren populazio-biomasarekiko. Ondorioz, indizearen selektibitatea 1en baliokidea zen adin eta urte guztietarako, eta 1ean finkatu zen SRA aplikatzerakoan. Urteko arrantza-jardueraren selektibitatea ezaguntzat jo zen, akatsik gabe, agertoki guztietan (3.2.Irudia), emaitzek arrantza-jardueraren selektibitate zehaztugabearekiko agertzen zuten sentikortasuna aztertzeko erabili ziren agertokietan izan ezik.

SRA eredua aplikatzerakoan, ziurgabetasun-parametroak "egiazko" balioetan ezarri ziren. Biomasa-indizearen datuen eta harrapaketen batez besteko luzera datuen bariazio-koefizientea 0,25ean finkatu zen, behaketa-ereduan, biomasa-indizearen datuak eta harrapaketen batez besteko luzera datuak sortzeko egin zen bezalaxe. Adin- eta luzera-edukiontzien banaketa luzeraren araberako harrapaketetan, adinaren araberako harrapaketetan eta adinaren araberako abundantzia-indizeari loturiko datuetan, behatutako datu hauek sortzeko erabili ziren lagin-tamaina bera erabiliz zehaztu zen. Luzera-adin bihurtzeko, adinaren araberako batez besteko luzeran 0,15eko bariazio-koefizientea ezarri zen, behaketa-ereduan erabili zen bezala.

Ikerketa honetan erabili zen SRA ereduaren bertsioa, xede berezi batekin diseinatu zen: harrapaketen datuek arrantza-jardueraren epealdi osoa bildu ez arren, populazioa ebaluatzeko gai izatea, alegia. Populazioa ustiatze-egoera batetik hasten denean, arrantzarik gabeko egoeran ematen den errekrutatzea eta hasierako harrapaketa-biomasa orekari dagokion harrapaketen heriotza-tasa kalkulatzen ditu, erabiltzaileak zehazten dituen hasierako harrapaketa-maila eta bizi-historia parametroak (populazio-errekrutatze erlazioa eta heriotza-tasa naturala, esaterako) oinarri hartuta. Hasierako oreka-egoera horretako harrapaketen heriotza-tasa horretatik abiatuta, hasierako murriztea kalkulatzen da (Huynh et al., 2020a). Populazioa arrantzarik gabeko egoera batetik hasten denean, hasierako murriztea ezaguna da (hau da, 0); eta parametro gutxiago daude kalkulatzeko. Horrenbestez, ebaluazio-eredua datuen lehen urtean arrantzarik gabeko egoeran dagoen populazio bati aplikatzea da egoerarik onena SRA eredurako eta, ziurrenik, baita beste edozein ebaluazio-eredu aplikatzerakoan ere, emaitzarik zehatzenak lortzeko (Huynh et al., 2020a,b).

Agertokiak

SRA ereduaren errendimendua 24 kasu ezberdinetan aztertu zen. 24 kasu horiek, sei datu mota eta lau epealdi-luzera ezberdinen konbinazioa dira. Agertoki guztietan, harrapaketa guztien datuak eta biomasa-indizearen datuak erabili ziren, aipatuak gutxieneko datuak baitira SRA eredua aplikatzeko. Datu motaren araberako agertoki batean ez genuen datu osagarririk sartu (CI agertokia); beste bost agertokietan, ondoko datu osagarriak sartu genituen: harrapaketen batez besteko luzeraren datuak (ML agertokia), luzeraren araberako harrapaketen datuak (CAL agertokia), adinaren araberako harrapaketen datuak (CAA agertokia), adinaren araberako abundantzia-indizearen datuak (IAA agertokia) eta adinaren araberako harrapaketen gehi adinaren araberako abundantzia-indizearen datuak (CAA+IAA agertokia). Aztertutako lau epealdi-luzera ezberdinek, 1978.urtea izan zuten abiapuntu, eta 2018an (41 urte), 2009an (32 urte), 1999an (22 urte) edo 1989an (12 urte) amaitu ziren. Epealdi-luzera ezberdinei loturiko agertokietan egon daitezkeen ezberdintasunak, horrenbestez, aztertutako epealdiaren luzeraren ondorio izango dira eta ez, ordea, datuen lehen urtean aintzat hartutako hasierako populazioaren ondorio.

Aipatutako datuen eskuragarritasunari loturiko agertokiak, hasierako bi populazio hipotesi ezberdinetarako aztertu ziren: (1) populazioa arrantzarik gabeko egoera batetik hasten da eta, ondorioz, harrapaketen datuak ustiatzearen hasiera-hasieratik ezagutzen direla onartzen da; eta (2) harrapaketen datuak arrantza-jarduera hasi eta ostekoak dira; horrenbestez, datuen lehen urtean populazioa ustiatze-baldintza ezezagun batean dago. Lehenengorako, ondoko populazioak erabili ziren: $\mathrm{OM}_{2\mathrm{F}_{\mathrm{MSY}}}$, $\mathrm{OM}_{\mathrm{F}_{\mathrm{MSY}}}$ eta $\mathrm{OM}_{0,5\mathrm{F}_{\mathrm{MSY}}}$. Bigarrenerako, legatzaren ebaluaziotik datorren harrapaketen heriotza-tasaren joeran oinarritutako populazioa erabili zen; hau da, $\mathrm{OM}_{\mathrm{F}_{\mathrm{SS3}}}$ populazioa. Legatzaren ebaluazioan (ICES, 2019a), 53.564 tonako harrapaketa-maila, lehen bost urtetako (1978tik 1982ra) batez besteko harrapaketa dena, erabiltzen da harrapaketa-biomasa orekari buruzko hipotesi gisa Stock Sintesi ebaluazio-eredua aplikatzen denean. Kapitulu honetan, 50.000 tonako harrapaketa-biomasa oreka erabiliz lorturiko SRA emaitzak arretaz aztertu ziren. Bestalde, 40.000-70.000 tona bitarteko harrapaketa-biomasa oreka ezberdinak erabiliz, emaitzek lehen urtean erabilitako harrapaketa mailarekiko agertzen zuten sentikortasuna ere aztertu zen.

Azkenik, arrantza-jardueraren selektibitate zehaztugabeak SRA ereduan duen eragina aztertu zen. Horretarako, $\mathrm{OM}_{\mathrm{F}_{\mathrm{SS3}}}$ populazioa eta 50.000 tonako harrapaketa-biomasa oreka erabili ziren eta datu-eskuragarritasunari dagozkien 24 agertokietarako selektibitate zehaztugabeak SRA ereduaren emaitzetan zuen sentikortasuna aztertu zen. Adinaren edo luzeraren araberako arrantza datuak ezaguntzat jo genituen agertoki horietan (hau da, CAL, CAA eta CAA+IAA agertokietan), hiru doikuntza ezberdin aztertu ziren selektibitatea zehazteko. Selektibitate-agertoki batean, selektibitateko hiru parametroak (L_5, L_{FS} eta V_{maxlen}) adinaren edo luzeraren araberako datuetatik abiatuta kalkulatu ziren (Kalkulatua agertokia). Beste bi selektibitate-agertokietan, L_5 eta L_{FS} parametroak adinaren edo luzeraren araberako datuetatik abiatuta kalkulatu ziren, baina V_{maxlen} parametroa, 3.2.Taulako balioetara eta 1era finkatu zen, azken horretan selektibitate kurbaren itxura logistikoa aintzat hartuz ($\mathrm{V}_{\mathrm{maxlen}}$ finkatua eta Logistikoa agertokiak, hurrenez hurren). Adinaren edo luzeraren araberako arrantza datuak ezezaguntzat jo genituen agertokietan (hau da, CI, ML eta IAA agertokietan) datu zehaztugabeko bi selektibitate-agertoki aztertu ziren. Lehen selektibitate-agertokian, selektibitate-kurbaren parametroak era uniformean hartu ziren “egiazko” selektibitatetik kalkulatutako balio-tarteetatik (3.2.Taula) (Lagidua agertokia). Bigarren selektibitate-agertokian, V_{maxlen} parametroa, 1ean finkatu zen eta beste bi selektibitate-parametroak era uniformean hartu ziren kalkulatutako balio-tartetik (3.2.Taula) (Logistikoa agertokia). Selektibitate-parametro bakoitzaren tartea %10eko bariazio-koefiziente bat aintzat hartuz zehaztu zen, kalkulatutako balioaren banaketa uniformean. Hiru selektibitate-parametroak epealdi-luzera bakoitzerako kalkulatu ziren, arrantza-jardueraren selektibitate-kurba, “egiazko” populazioan ikusitako kurbetara egokituz.

Taula 3.2: "Egiazko" populazioa oinarri hartuta kalkulatutako (Kal) arrantza-jardueraren selektibitate-parametroak eta kalkulu horietako muga minimoa (min) eta maximoa (max) %10eko bariazio-koefizientedun banaketa uniforme batean. Selektibitate-parametroak: selektibitatearen %5aren luzera (L_5), selektibitate osoaren luzera (L_{FS}) eta selektibitatea luzera asintotikoan (V_{maxlen}). Epealdi-luzera (TS): 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

	Selektibitate-parametroak								
	L_5			L_{FS}			V_{maxlen}		
TS	**Min**	**Kal**	**Max**	**Min**	**Kal**	**Max**	**Min**	**Kal**	**Max**
41	10,77	13,03	15,29	53,38	64,56	75,74	0,27	0,33	0,39
32	3,99	4,82	5,66	49,66	60,07	70,47	0,25	0,31	0,36
22	0,48	0,58	0,68	41,10	49,71	58,32	0,28	0,34	0,40
12	4,85	5,86	6,88	21,25	25,70	30,15	0,31	0,37	0,44

3.3.Taulak agertoki bakoitzean kalkulatutako parametroen zerrenda laburbiltzen du.

Taula 3.3: Selektibitatearen (Sel), populazioaren (Pop), datu motaren (DT), epealdi-luzeraren (TS) eta harrapaketa-biomasa orekari buruzko hipotesiaren (Feq) agertoki-konbinazioaren arabera kalkulatutako parametroak. Selektibitate-agertokiak: urteroko selektibitatea "egiazko" balioetan (Egiazkoa) finkatu zen, selektibitatea luzera asintotikoan balio optimoran finkatu zen ($\mathrm{V_{maxlen}}$ finkatua), selektibitate-parametroak kalkulatu egin ziren (Kalkulatua), selektibitate logistikoa ezarri zen (Logistikoa) eta selektibitate-parametroak era uniformean hartu ziren balio-tarte zehatz bat oinarri hartuta (Lagindua). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

					Kalkulatutako parametroak							
Agertokiak					F_y	Sel param			R_0	δ_y	Feq	q
Sel	Pop	DT	TS	Feq		L_5	L_{FS}	V_{maxlen}				
Egiazkoa	$\mathrm{OM_{2F_{MSY}}}$	CI,ML,CAL, CAA,IAA, CAA+IAA	41, 32, 22, 12	Arrantzarik gabeko egoera	x				x	x		x
	$\mathrm{OM_{F_{MSY}}}$				x				x	x		x
	$\mathrm{OM_{0,5F_{MSY}}}$				x				x	x		x
	$\mathrm{OM_{F_{SS3}}}$			40.000-70.000	x				x	x	x	x
$\mathrm{V_{maxlen}}$ finkatua		CAL,CAA, CAA+IAA		50.000	x	x	x		x	x	x	x
Kalkulatua					x	x	x	x	x	x	x	x
Logistikoa					x	x	x		x	x	x	x
		CI,ML,IAA			x				x	x	x	x
Lagindua					x				x	x	x	x

3.2.2 Kalkulu nagusiak

Datu mota ezberdinek, epealdi-luzera ezberdinek eta hasierako populazioari buruzko hipotesi ezberdinek emaitzetan duten eragina ebaluatzeko, lau datu kalkulatu ziren (aurrerantzean, "kalkulu nagusiak"). Kalkulu horiek kudeaketarako duten interes potentzialagatik aukeratu ziren: murriztea (D, bat ken aurreko urteko biomasa ugaltzailearen eta arrantzarik gabeko egoeran dagoen biomasa ugaltzailearen arteko ratioa), arrantzarik gabeko errekrutatzea (R_0), azken urteko biomasa ugaltzailea ($\mathrm{SSB}_{\mathrm{LY}}$) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Horrez gain, hasierako populazioari buruzko hipotesiak emaitzetan duen eragina ere aztertu zen. Horretarako, hasierako murriztearen ($\mathrm{D}_{\mathrm{init}}$) eta SSB kalkuluak aztertu ziren. Hasierako murriztearen balioak, 0 eta 1en artean daude: 0 k populazioa arrantzarik gabeko egoeran dagoela adierazten du, eta 1 ek esan nahi du populazioa desagertua dagoela.

3.2.3 Errendimendu-estatistikak

SRA ereduaren errendimendua ebaluatzeko, kalkulu nagusien zehaztasun eta alborapen estatistikak erabili ziren (Ono et al., 2014; Walther and Moore, 2005). Errendimendua ebaluatzeko, SRA eredu bateratuak emandako kalkuluak aukeratu ziren bakarrik. Populazio-erreplika guztietako medianak erabili ziren batez besteko balioen ordez, errendimendu-estatistiken balio atipikoak saihesteko.

i populazio-erreplika bakoitzerako, θ_i-k kalkulu nagusien "egiazko" balioa OMn, eta $\widehat{\theta}_i$-k SRA eredutik kalkulatutako balioa (hau da, SRA ereduak emandako 100 kalkuluren mediana) adierazten dute. Kalkulu bakoitzaren zehaztasuna neurtzeko aintzat hartu ziren medianaren akats absolutu erlatiboa (MARE) eta akats-ratioa (FR). Medianaren akats absolutu erlatiboa (MARE), populazio-erreplika guztietan zehar ematen den akats erlatibo absolutuaren (ARE) mediana da, eta honela kalkulatu zen:

$$MARE = \ Median \ \left(\left| \frac{\widehat{\theta}_1 - \theta_1}{\theta_1} \right|, \ \ldots, \left| \frac{\widehat{\theta}_{100} - \ \theta_{100}}{\theta_{100}} \right| \ \right) * 100, \tag{3.7}$$

non zenbat eta handiagoa izan MARE balioa, are eta baxuagoa den kalkulu nagusiaren zehaztasuna. MARE balioek, kalkulu nagusien zehaztasunari buruzko ezberdintasun erlatiboa aztertzea ahalbidetzen dute.

FRa, Magnusson and Hilborn (2007)en bezalaxe, matematikoki, "egiazko" balioaren erdiaren behetik edo bikoitzetik gora zeuden kalkuluen proportzio gisa zehaztu zen.

$$FR \ = Pr \left(\left| \widehat{\theta}/\theta \right| \ < \ 0,5 \ and \ \left| \widehat{\theta}/\theta \right| \ > 2 \right). \tag{3.8}$$

"Egiazko" balioaren erdiaren behetik edo bikoitzetik gora dauden kalkuluak, gehienetan, detektaezinak dira benetako egoera batean. Errendimendu-estatistika honek, FRk, emaitzak ebaluatzea ahalbidetzen digu, akats nahiko baxua detektatzea ahalbidetuz kalkulu nagusien zehaztasunean.

Alborapena kalkulatzeko, populazio-erreplika guztien medianaren akats erlatiboa (MRE) erabili zen, jarraian adierazi bezala:

$$MRE = \ Median \ \left(\frac{\widehat{\theta}_1 - \ \theta_1.}{\theta_1}, \ \ldots, \frac{\widehat{\theta}_{100} - \ \theta_{100}}{\theta_{100}} \right) * 100, \tag{3.9}$$

non zenbat eta handiagoa izan MREren balio absolutua, are eta alborapen handiagoa egongo den kalkuluetan.

3.3 Emaitzak

Orokorrean, SRA ereduaren bateratze-indizea %80tik gorakoa izan zen eta %100era iritsi zen kasu gehienetan. Nabarmendu beharra dago, bateratze-indizea %17 ingurukoa izan zela harrapaketen batez besteko luzeraren datuak erabili zirenean (ML agertokia) datu epealdi luzeenarekin eta zehaztu gabeko selektibitatearekin (Lagindua eta Logistikoa agertokietan). E.3 Eranskineko E.1.Taulak, agertoki bakoitzeko bateratze-indizeak jasotzen ditu.

3.3.1 Arrantzarik gabeko egoera

Kalkulu nagusiak zehatzagoak ziren (hau da, FR eta MARE baxuagok) epealdi luzeago bat erabili zenean. Datu motaren araberako agertoki guztietan kalkulu nagusien FR eta MARE balioek gora egin zuten epealdiaren luzera murriztu zenean (3.4.Taula, 3.3.Irudia). Akats erlatibo absolutuaren (ARE) tartea ere handiagoa zen epealdiaren luzera murrizten zenean (E.4 Eranskineko E.2.Irudia). Kalkulu nagusien zehaztasuna handia zen (FR %20tik beherakoa) epealdi-luzera eta datu mota konbinazio guztien kasuan, epealdi motzenaren (12 urte) kasuan ezik. Azken horretan, FRak %36ra arteko balioak hartu zituen eta MAREk, %50era artekoak.

CI agertokian, beste agertokietan baino datu gutxiago erabili ziren eta epealdi-luzera guztietan, D, R_0, SSB_{LY} eta F_{LY} kalkuluek balio kaskarrenak agertu zituzten zehaztasunari dagokionez (CI agertokiak FR eta MARE balio altuenak agertzen zituen) (3.4.Taula). Kalkulu nagusien zehaztasuna handiagoa zen luzeraren araberako harrapaketen datuak erabiliz gero (CAL agertokia), harrapaketen batez besteko luzeraren datuak erabili ordez (ML agertokia). MARE adierazlea baxuagoa zen CAL agertokian ML agertokian baino. Kalkulu nagusien zehaztasuna antzekoa zen luzeraren araberako harrapaketak, adinaren araberako harrapaketak edo adinaren araberako abundantzia-indizearen datuak erabili ziren agertokietan (CAL, CAA eta IAA agertokietan, hurrenez hurren). Oro har, kalkulu nagusi zehatzenak daturik osatuenak erabili zirenean lortu ziren (CAA eta IAA agertokietan).

Epealdi luzera handitzen zenean, SSB_{LY} eta F_{LY} kalkuluen zehaztasuna D eta R_0 kalkuluen zehaztasuna baino gehiago handitzen zen (3.4.Taula). Kasu guztietan, D eta R_0 ren kalkuluak SSB_{LY} eta F_{LY} ren kalkuluak baino zehatzagoak ziren (hau da, FR eta MARE baxuagoak zituzten). F_{LY} balioak zuen zehaztasunik baxuena. CI agertokian, kalkulu nagusirik zehatzena D zen. ML, CAL, CAA, IAA eta CAA+IAA agertokietan, MARE baliorik baxuena R_0k zuen, epealdi motzenaren kasuan ezik. Azken horretan, D ren zehaztasuna R_0 rena baino altuagoa zen. D eta R_0 kalkuluen alborapena baxuagoa zen SSB_{LY} eta F_{LY} kalkuluena baino (hau da, RE balio absolutu txikiagoak) (3.4.Taula, eta E.4 Eranskineko E.3.Irudia).

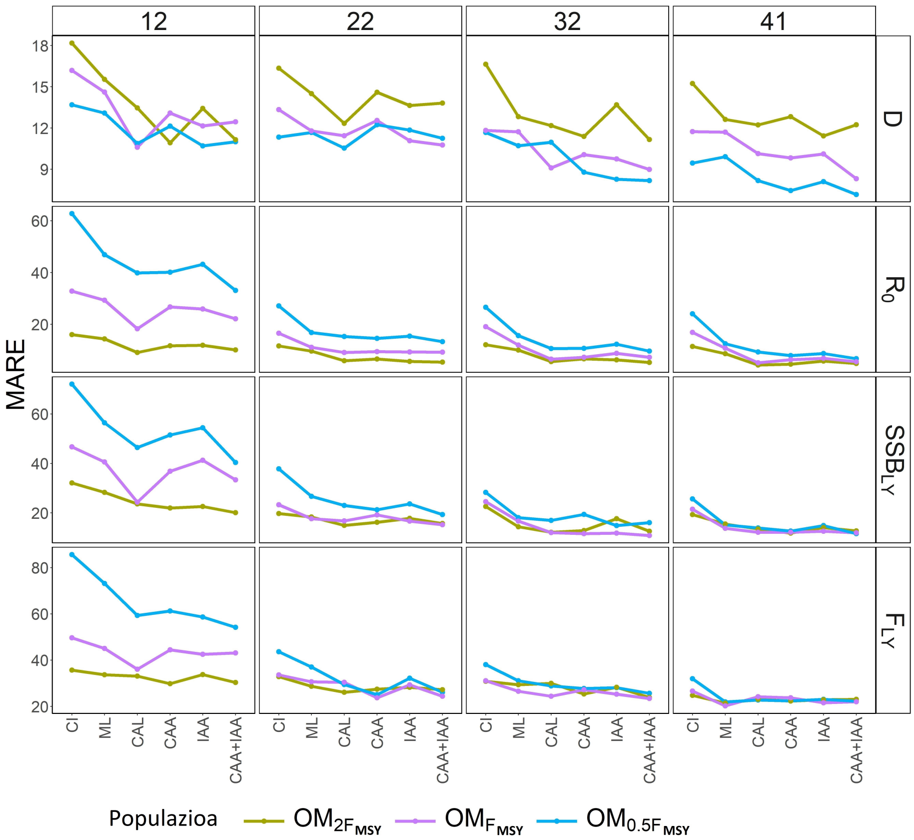
Kalkulu nagusien zehaztasuna populazioen harrapaketen heriotza-tasaren menpe zegoen (3.3.Irudia). R_0, SSB_{LY} eta F_{LY} kalkuluen zehaztasunak gora egiten zuen populazioan harrapaketen heriotza-tasa handiagoa zenean. Kalkulu nagusi horiek MARE baxuagoa zuten $OM_{2F_{MSY}}$ populazioan, $OM_{F_{MSY}}$ populazioarekin alderatuta; eta MARE altuagoa zuten $OM_{0,5F_{MSY}}$ populazioan, $OM_{F_{MSY}}$ populazioarekin alderatuta (3.3.Irudia). D kalkuluek kontrako joera agertu zuten: D ren zehaztasuna baxuagoa zen $OM_{2F_{MSY}}$ populazioan eta altuagoa $OM_{0,5F_{MSY}}$ populazioan, $OM_{F_{MSY}}$

populazioarekin alderatuta. Zenbat eta laburragoa izan epealdi-luzera, are eta alde handiagoa zegoen populazioen artean R_0, SSB_{LY} eta F_{LY} kalkuluen zehaztasunari dagokionez. Eta, aitzitik, D ren MARE balioen arteko aldea $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ eta $OM_{0,5F_{MSY}}$ populazioetan txikiagoa zen epealdi-luzerarik laburrena aintzat hartzen zenean (3.3.Irudia).

Taula 3.4: Kalkulu nagusien errendimendu-estatistikak datu mota eta epealdi-luzera agertoki-konbinazio bakoitzerako, $OM_{F_{MSY}}$ populazioa erabili zenean. Errendimendu-estatistikak: akats-ratioa (FR), medianaren akats erlatibo absolutua (MARE), medianaren akats erlatiboa (MRE). Kalkulu nagusiak: murriztea (D), errekrutatzea arrantzarik gabeko egoeran (R_0), azken urteko biomasa ugaltzailea (SSB_{LY}) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

		FR				**MARE**				**MRE**			
		12	**22**	**32**	**41**	**12**	**22**	**32**	**41**	**12**	**22**	**32**	**41**
D	**CI**	0	0	0	0	16	13	12	12	0	-1	4	1
	ML	0	0	0	0	15	12	12	12	-3	-7	0	-4
	CAL	0	0	0	0	11	11	9	10	-5	-7	-3	-7
	CAA	0	0	0	0	13	13	10	10	-1	0	3	0
	IAA	0	0	0	0	12	11	10	10	-1	-1	0	0
	CAA+IAA	0	0	0	0	12	11	9	8	-2	1	1	-1
$\mathbf{R_0}$	**CI**	20	5	0	0	33	17	19	17	27	15	19	17
	ML	12	0	0	0	29	11	12	11	7	8	11	11
	CAL	0	0	0	0	18	9	7	5	-12	-7	-4	-3
	CAA	13	0	0	0	27	9	7	6	3	5	5	5
	IAA	7	0	0	0	26	9	9	7	6	5	7	5
	CAA+IAA	8	0	0	0	22	9	7	6	4	5	5	4
$\mathbf{SSB_{LY}}$	**CI**	31	15	4	2	47	23	25	21	25	16	21	16
	ML	24	3	0	0	41	18	17	14	8	3	7	7
	CAL	11	0	0	0	24	17	12	12	-16	-13	-7	-8
	CAA	21	1	0	0	37	19	12	12	3	4	7	6
	IAA	20	0	2	0	41	17	12	13	1	4	6	4
	CAA+IAA	20	0	0	0	33	15	11	12	2	4	4	4
$\mathbf{F_{LY}}$	**CI**	36	19	17	7	50	34	31	27	-23	-16	-17	-17
	ML	32	12	10	3	45	31	27	20	-10	-8	-7	-9
	CAL	21	10	6	2	36	30	24	24	15	12	8	8
	CAA	34	8	8	4	44	24	27	24	-13	-4	-5	-7
	IAA	28	6	10	4	43	29	25	22	-11	-4	-9	-4
	CAA+IAA	27	4	9	5	43	24	23	22	-7	2	-6	-3

Oro har, D ren kalkuluak R_0 ren kalkuluak baino zehatzagoak ziren azken urteotako ustiatze-maila baxua zenean ($OM_{0,5F_{MSY}}$ populazioan); eta R_0 ren kalkuluak D ren kalkuluak baino zehatzagoak ziren populazioa oso ustiatuta zegoenean ($OM_{2F_{MSY}}$ populazioan) (3.3.Irudia).



Irudia 3.3: Kalkulu nagusien (errenkadak) medianaren akats erlatibo absolutua (MARE) datu mota (x-ardatza) eta epealdi-luzera (zutabeak) agertoki-konbinazio bakoitzerako $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ eta $OM_{0,5F_{MSY}}$ populazioak erabili zirenean. Kalkulu nagusiak: murriztea (D), errekrutatzea arrantzarik gabeko egoeran (R_0), azken urteko biomasa ugaltzailea (SSB_{LY}) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

3.3.2 Ustiatze-egoera

$OM_{F_{SS3}}$ populazioa erabili zenean ere, epealdirik luzeenak eman zituen kalkulu zehatzenak. Errendimendurik txarrena, epealdi laburrenarekin lortu zen (3.5.Taula, eta E.5 Eranskineko E.4.Irudia eta E.5.Irudia). Epealdi-luzerak eragin handia zuen kalkulu nagusien zehaztasunean beharrezko gutxieneko datuak erabili zirenean (CI agertokia) (3.5.Taula). CI agertokiak kalkulu nagusi zehatzak eman zituen, alborapenik gabekoak (hau da, %10etik beherako FR eta -8 eta 6 bitarteko MRE), epealdi luzeena erabili zenean (41 urte). Epealdi laburragoekin (hau da, 32, 22 eta 12 urteko epealdien datuak erabili zirenean), ordea, D, SSB_{LY} eta F_{LY} kalkuluen FR %40tik gorakoa izan zen. Hau da, simulazioen %40ak baino gehiagok, kalkulu nagusien balioa “egiazko” balioa baino bi aldiz handiagoa edo bi aldiz txikiagoa kalkulatu zuten. Datu osagarriekin (hau da, ML, CAL, CAA, IAA eta CAA+IAA agertokietan), epealdi-luzera guztietan, kalkulu nagusien zehaztasuna altua izan zen (kalkulu nagusien FR %20tik beherakoa izan zen). Datu motari loturiko agertoki guztietan, kalkuluen alborapenaren mediana 0 tik gertu zegoen (3.5.Taula) eta kalkulu nagusien "egiazko" balioa akats erlatiboaren (RE) %80ko konfiantza-tartearen barruan zegoen (E.5 Eranskineko E.6.Irudia).

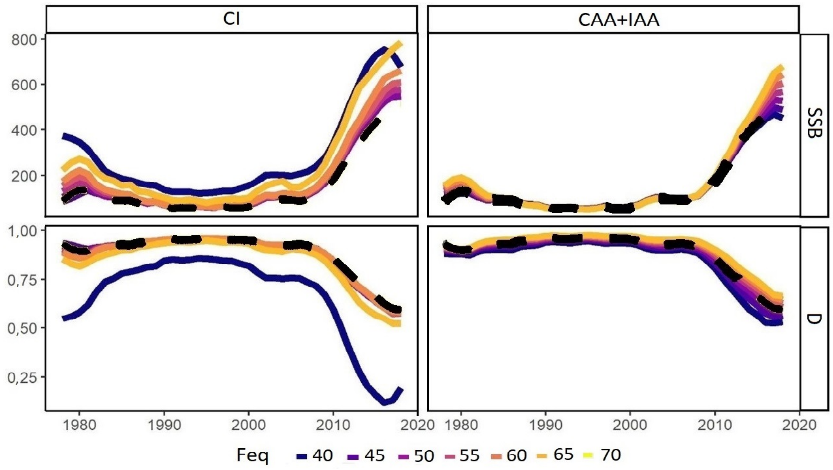
Taula 3.5: Kalkulu nagusien errendimendu-estatistikak datu mota eta epealdi-luzera agertoki-konbinazio bakoitzerako, $OM_{F_{SS3}}$ populazioa erabili zenean 50.000 tonako harrapaketa-biomasa orekari buruzko hipotesiarekin. Errendimendu-estatistikak: akats-ratioa (FR), medianaren akats erlatibo absolutua (MARE), medianaren akats erlatiboa (MRE). Kalkulu nagusiak: murriztea (D), errekrutatzea arrantzarik gabeko egoeran (R_0), azken urteko biomasa ugaltzailea (SSB_{LY}) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

		FR				**MARE**				**MRE**			
		12	**22**	**32**	**41**	**12**	**22**	**32**	**41**	**12**	**22**	**32**	**41**
D	**CI**	72	70	62	4	325	203	130	24	325	203	130	3
	ML	14	8	3	0	31	29	17	17	-10	-15	-15	-11
	CAL	0	0	0	0	19	18	13	12	-7	-7	-8	-5
	CAA	3	1	0	0	23	18	15	12	13	7	3	1
	IAA	1	1	0	0	21	19	16	10	9	9	3	2
	CAA+IAA	0	1	0	0	19	17	14	11	9	6	2	3
R_0	**CI**	18	16	7	0	25	25	24	13	-11	-20	-21	6
	ML	3	2	0	0	19	19	18	15	9	14	17	15
	CAL	0	0	0	0	9	10	9	8	1	4	4	3
	CAA	0	0	0	0	12	9	7	6	-3	-2	-2	0
	IAA	0	0	0	0	9	9	8	7	-1	1	1	1
	CAA+IAA	0	0	0	0	11	8	7	6	-4	-1	-1	0
SSB_{LY}	**CI**	64	56	42	1	225	138	76	13	225	138	76	5
	ML	7	2	1	0	21	19	14	12	5	-2	-1	0
	CAL	0	0	0	0	17	14	12	11	-2	-3	-2	-1
	CAA	0	0	0	0	18	15	15	9	13	5	4	2
	IAA	1	0	0	0	21	18	15	10	5	7	5	3
	CAA+IAA	0	0	0	0	17	15	14	11	8	5	6	2
F_{LY}	**CI**	66	57	56	3	77	63	55	24	-72	-61	-52	-8
	ML	20	15	13	2	29	33	27	19	-15	1	1	-2
	CAL	17	12	11	2	31	30	26	21	-5	8	-2	-3
	CAA	15	13	10	1	32	34	25	20	-22	-3	-8	-2
	IAA	14	12	10	1	35	34	25	20	-16	-6	-10	-5
	CAA+IAA	13	13	9	1	30	34	23	20	-18	-6	-6	-4

D, SSB_{LY} eta F_{LY} kalkuluen zehaztasuna antzekoagoa zen, harrapaketen batez besteko luzeraren datuak erabili ziren agertokian (ML agertokia) eta luzeraren araberako datuak edo adinaren araberako datuak erabili ziren agertokietan (CAL, CAA, IAA eta CAA+IAA agertokiak), R_0 kalkuluaren zehaztasuna baino. R_0 ren MARE balioak bi aldiz inguru handiagoak ziren harrapaketen batez besteko luzeraren datuak erabili zirenean, luzeraren araberako edo adinaren araberako datuak erabili zirenean baino (3.5.Taula). Gauza bera gertatu zen alborapenarekin: R_0 kalkuluaren alborapena lau aldiz handiagoa zen ML agertokian (R_0 ren MRE balio absolutua lau aldiz inguru handiagoa zen), luzeraren araberako edo adinaren araberako datuak erabili ziren agertokietan baino (3.5.Taula).

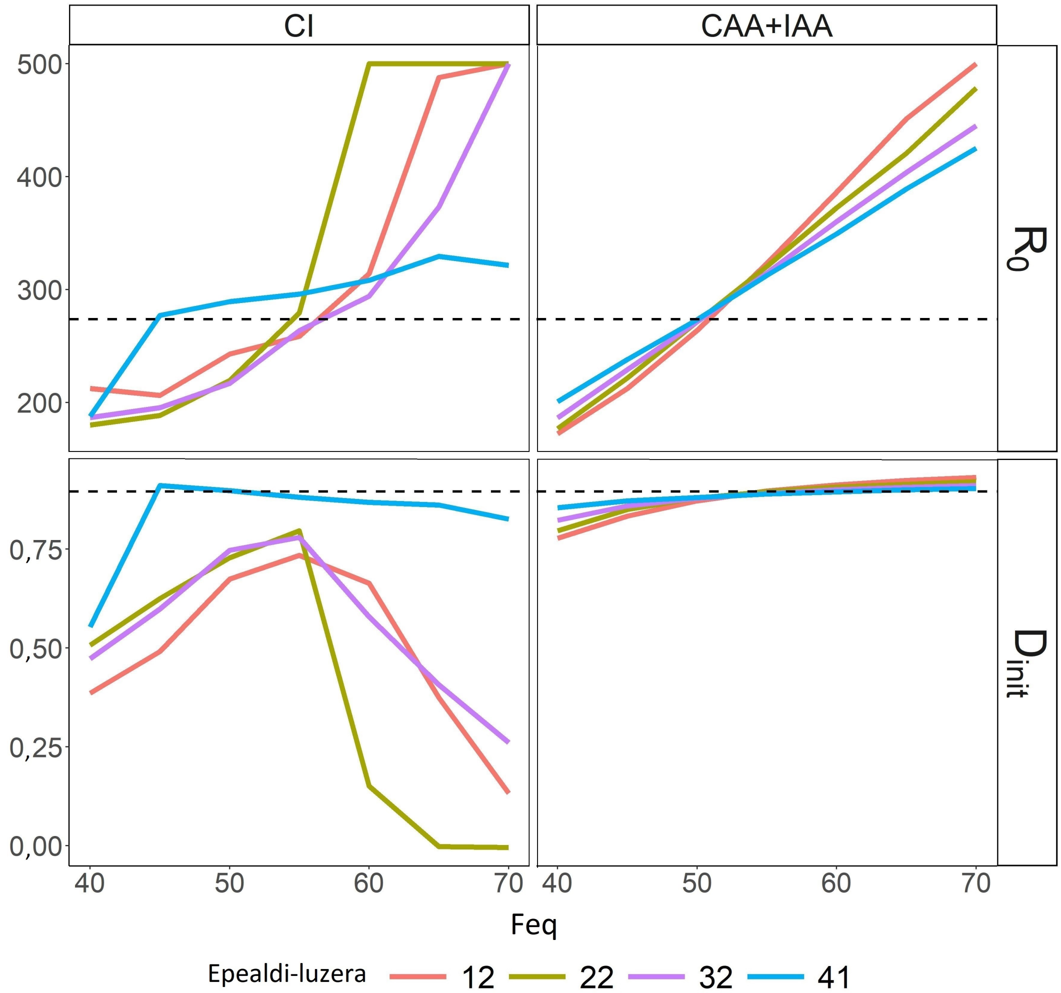
3.3.2.1 Harrapaketa-biomasa orekari buruzko hipotesiarekiko kalkulu nagusiek agertzen duten sentikortasunaren analisia

Datuen eskuragarritasun ezberdinetako agertoki guztietan, R_0, SSB, D eta D_{init} kalkuluen balioek gora egin zuten harrapaketa-biomasa orekari buruzko hipotesi gisa harrapaketa-maila handiagoak erabili zirenean (3.4.Irudia, 3.5.Irudia). Salbuespen gisa, beharrezko gutxieneko datuak (CI agertokia) eta epealdirik luzeena (41 urte) erabili zirenean, harrapaketa-biomasa orekari buruzko hipotesi gisa harrapaketa baxuenak erabiltzean, harrapaketa altuenak erabiltzean baina SSB kalkulu altuagoak lortu ziren (3.4.Irudia). Hori gertatu zen, harrapaketa-baxuenek, hasierako murrizte baxua ere ekarri zuelako (3.5.Irudia). Harrapaketa-biomasa orekaren maila handitu ahala, R_0-ren kalkulu balioak azkarrago handitu ziren, beste kalkulu nagusien balioak baino (3.6.Irudia), azken horiek aldatzeko joera txikiagoa agertzen baitzuten harrapaketa-biomasa oreka maila ezberdinen aurrean. Beharrezko gutxieneko datuak erabili zirenean, kalkulu nagusien zehaztasunak aldaketa irregularragoak agertzen zituen harrapaketa-biomasa orekaren maila aldatzen zenean, beste datu osagarri batzuk ere erabiltzen zirenean baino (3.5.Irudia, 3.6.Irudia). Oro har, kalkuluen zehaztasuna altuagoa zen harrapaketa-biomasa orekaren maila-tarte zabalago batean epealdi luzeagoak erabili zirenean, epealdi laburragoak erabili zirenean baino (3.6.Irudia).

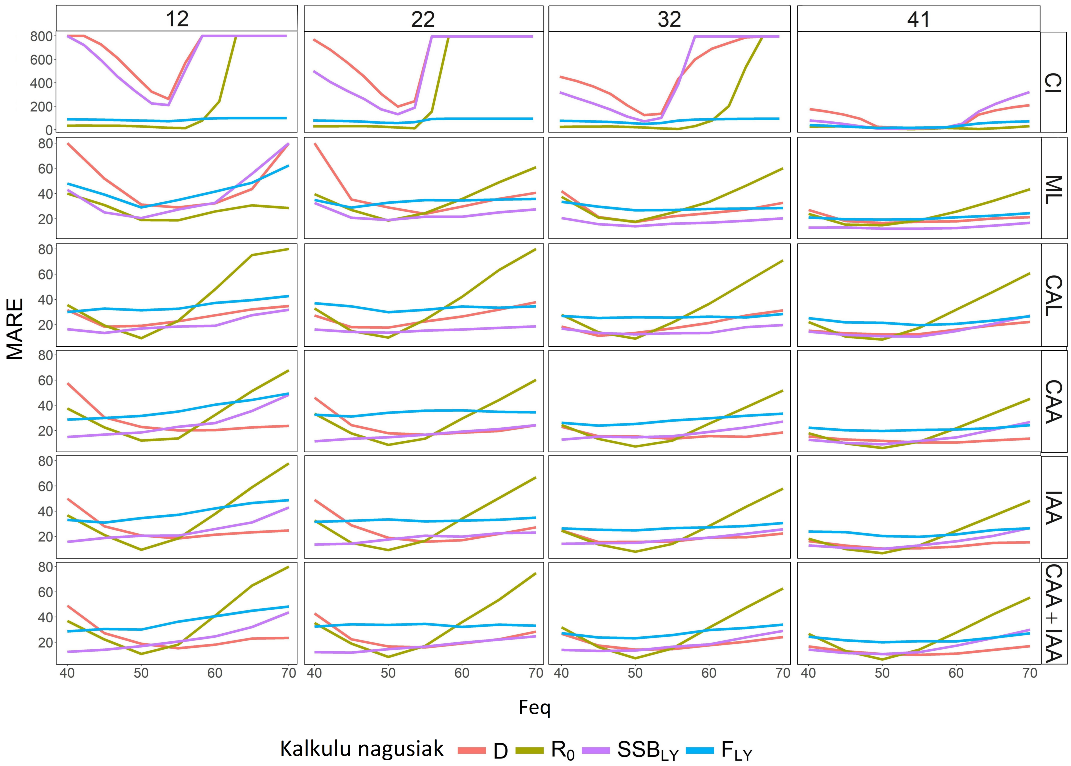


Irudia 3.4: Biomasa ugaltzailearen (SSB, 1.000 tonaka) eta murriztearen (D) kalkuluak harrapaketa-biomasa orekari buruzko hipotesi ezberdinekiko (Feq, 1.000 tonaka) 41 urteko epealdia erabili zenean harrapaketa eta biomasa-indizearen datuekin (CI agertokia) eta CI gehi adinaren araberako harrapaketa eta abundantzia-indizearen datuekin (CAA+IAA agertokia) $OM_{F_{SS3}}$ populazioan. Lerro jarraituek harrapaketa-biomasa orekari buruzko hipotesi bakoitzerako kalkuluak irudikatzen dituzte eta lerro eten beltzek “egiazko” populazio-errepliken mediana irudikatzen dute.

Datu mota bakoitzak eskaintzen duen informazioaren arabera, eta datuen epealdi-luzeraren arabera, kalkulu zehatzenak lortzea ahalbidetzen zuen harrapaketa-biomasa orekari buruzko hipotesirik egokiena ezberdina zen (3.5.Irudia, 3.6.Irudia). Halaber, kalkulu nagusi bakoitzaren baliorik zehatzena, harrapaketa-biomasa orekari buruzko hipotesi ezberdinetan lortzen zen (3.5.Irudia, 3.6.Irudia). Harrapaketa-biomasa orekari buruzko hipotesiak, SSB ren lehen eta azken zazpi urteetako kalkuluetan eragiten zuen gehien bat (3.4.Irudia). SSB ri loturiko kalkuluak berdintsu mantendu ziren epealdi-luzeraren erdialdean harrapaketa-biomasa orekari buruzko hipotesia zena zela, eta tarte horretan, SSB ri loturiko kalkuluak "egiazko" populazioaren SSB balioetatik gertu zeuden datu mota agertokirik osatuena erabili zenean (3.4.Irudia).



Irudia 3.5: Arrantzarik gabeko errekrutatzearen (R_0 milioitan) eta hasierako murriztearen (D_{init}) kalkuluen medianak (errenkadak) harrapaketa-biomasa orekari buruzko hipotesi ezberdinak (Feq, 1.000 tonaka) (x-ardatza), epealdi-luzera ezberdinak eta datu mota agertokirik oinarrizkoena (CI agertokia) eta osatuena (CAA+IAA agertokia) (zutabeak) erabili zirenean $OM_{F_{SS3}}$ populazioarekin. Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta adinaren araberako harrapaketen datuak eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra). Lerro eten horizontal beltzek irudikatzen dute "egiazko" populazioaren R_0 eta D_{init} balioen mediana.

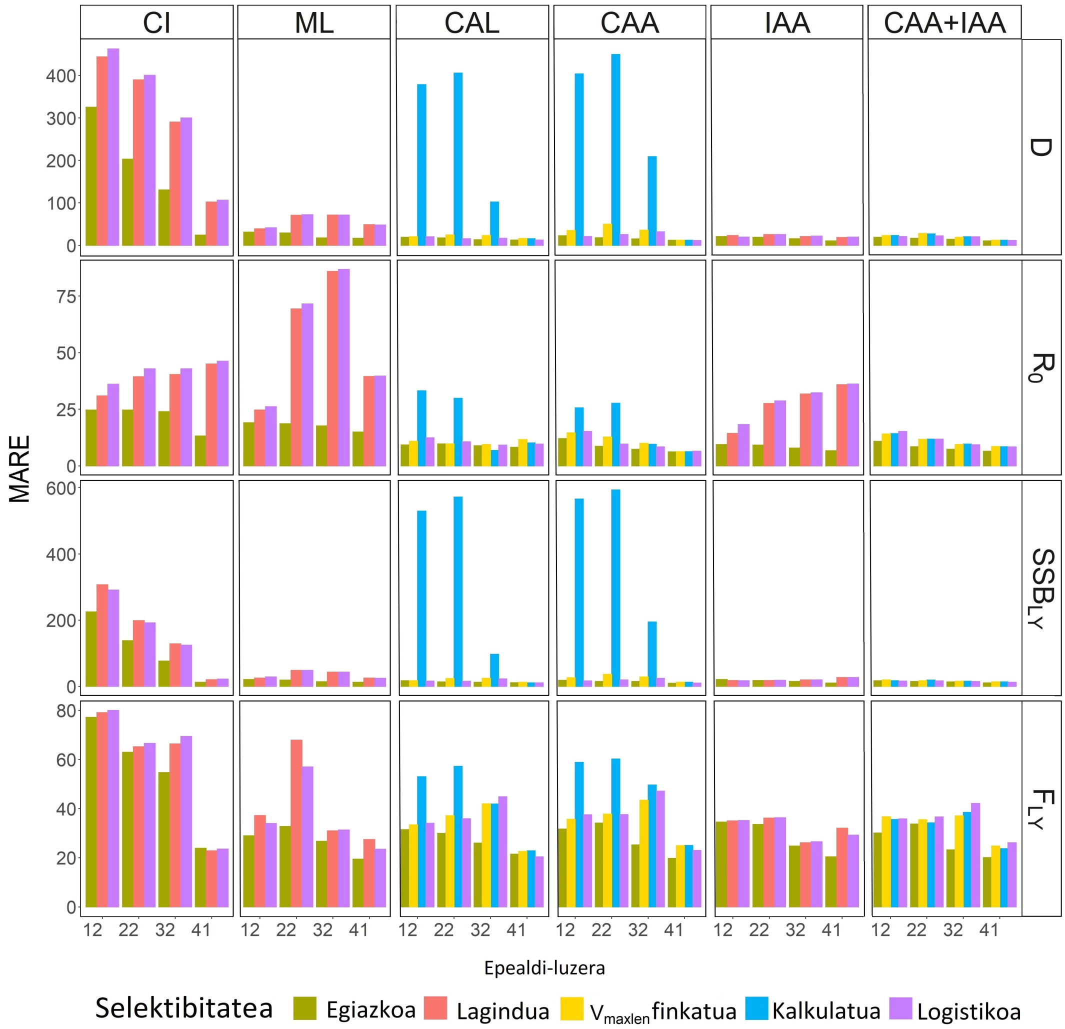


Irudia 3.6: Kalkulu nagusien medianaren akats erlatibo absolutua (MARE) harrapaketa-biomasa orekari buruzko hipotesi ezberdinetan (Feq, 1.000 tonaka), datu mota (errenkadak) eta epealdi-luzera (zutabeak) agertoki-konbinazio bakoitzerako, $OM_{F_{SS3}}$ populazioa erabili zenean. Kalkulu nagusiak: murriztea (D), errekrutatzea arrantzarik gabeko egoeran (R_0), azken urteko biomasa ugaltzailea (SSB_{LY}) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra). CI datu motaren y-ardatza 0tik 800era arte doa; beste datu mota guztien y-ardatza, aitzitik, 0tik 80ra arte. CI agertokian, 800tik gorako MARE balioak 800en finkatu ziren.

3.3.2.2 Kalkulu nagusiek arrantza-jardueraren selektibitatearen doikuntza ezberdinekiko agertzen duten sentikortasunaren analisia

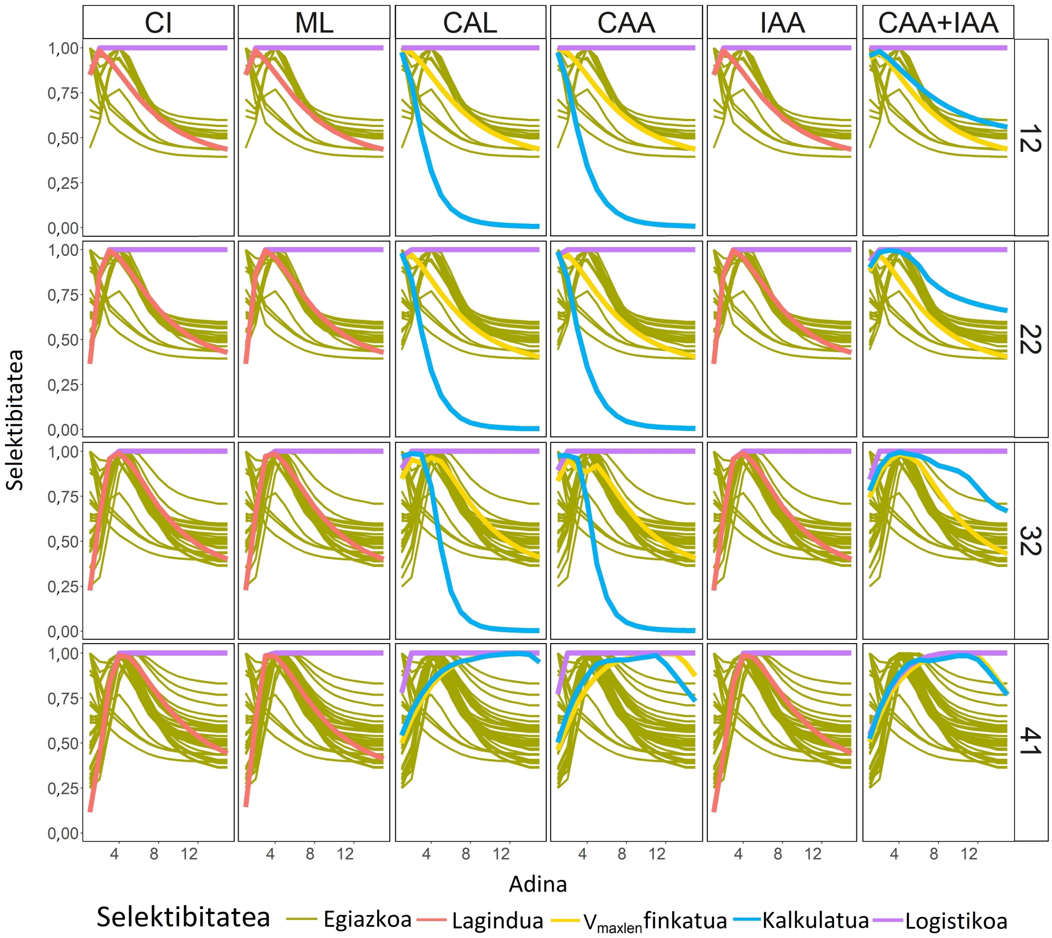
Espero zitekeen bezala, kalkulu nagusi zehatzenak, arrantza-jardueraren selektibitaterako "egiazko" populazioaren balioak erabili zirenean lortu ziren (Egiazkoa agertokia) (3.7.Irudia). Adinaren araberako harrapaketen eta abundantzia-indizearen datuak erabili zirenean (CAA+IAA agertokia), selektibitate-agertoki ezberdinekin lorturiko kalkulu nagusien zehaztasuna antzekoa zen (3.7.Irudia). Baina adinaren araberako harrapaketen datuak edo luzeraren araberako harrapaketen datuak erabili zirenean selektibitatearen parametro guztiak kalkulatzeko (Kalkulatua agertokia), adinaren araberako abundantzia-indizearen datuak erabili gabe (hau da, CAA eta CAL agertokiak erabili zirenean), banako zaharrenen selektibitatea gutxietsi egin zen (3.7.Irudia) eta gehiegizko populazioaren biomasa kalkulatu zen (E.5 Eranskineko E.7.Irudia). Agertoki honetan, D eta SSB_{LY} kalkuluen zehaztasuna baxua zen, beste selektibitate-agertoki guztiekin alderatuta (3.7.Irudia). Kalkulu nagusien zehaztasuna handiagoa zen luzera asintotikoko selektibitatea balio optimoan zehazten zenean

(V_{maxlen} finkatua agertokia) edo itxura logistikora egokitzen zenean (Logistikoa agertokia), selektibitate-parametroa kalkulatzen zenean baino (Kalkulatua agertokia). Kalkulu nagusien zehaztasuna, selektibitate-kurbaren beheranzko adarra finkatu zen bi agertokietan antzekoa zen (3.7.Irudia).



Irudia 3.7: Kalkulu nagusien (errenkadak) medianaren akats erlatibo absolutua (MARE) datu mota (zutabeak) eta epealdi-luzera (x-ardatza) agertoki-konbinazio bakoitzerako bost selektibitate agertoki ezberdinetan, $OM_{F_{SS3}}$ populazioa eta harrapaketa-biomasa orekari buruzko hipotesia 50.000 tonatan finkatu zenean. Kalkulu nagusiak: murriztea (D), errekrutatzea arrantzarik gabeko egoeran (R_0), azken urteko biomasa ugaltzailea (SSB_{LY}) eta azken urteko harrapaketen heriotza-tasa (F_{LY}). Selektibitate-agertokiak: urteroko selektibitatea "egiazko" balioetan finkatu zen (Egiazkoa), selektibitate-parametroak balio-tarte batetik aukeratu ziren era uniformean (Lagindua), selektibitatearen luzera asintotikoa balio optimoan finkatu zen (V_{maxlen} finkatua), selektibitate-parametroak kalkulatu egin ziren (Kalkulatua), selektibitatearen luzera asintotikoa 1en finkatu zen selektibitate logistikoa onartuz (Logistikoa). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra).

Erabilitako selektibitate-kurbak, ezberdinak ziren selektibitatearen parametro balioak 3.2.Taulako balio-tartetik zehaztu zirenean (Lagindua agertokia) eta beheranzko adarren forma logistikoa zela onartu zenean (Logistikoa agertokia) (3.8.Irudia). Hala ere, kalkulu nagusien zehaztasuna, antzekoa zen bi agertoki horietan (3.7.Irudia).



Irudia 3.8: Adinaren araberako selektibitatea, selektibitatearen bost agertokietan datu mota (zutabeak) eta epealdi-luzera (errenkadak) agertoki-konbinazio bakoitzerako, $OM_{F_{SS3}}$ populazioa eta harrapaketa-biomasa orekari buruzko hipotesia 50.000 tonatan finkatu zenean. Selektibitate-agertokiak: urteroko selektibitatea "egiazko" balioetan finkatu zen (Egiazkoa), selektibitate-parametroak balio-tarte batetik aukeratu ziren era uniformean (Lagindua), selektibitatearen luzera asintotikoa balio optimoan finkatu zen (V_{maxlen} finkatua), selektibitate-parametroak kalkulatu egin ziren (Kalkulatua), selektibitatearen luzera asintotikoa 1en finkatu zen selektibitate logistikoa onartuz (Logistikoa). Datu moten agertokiak: harrapaketen eta indizearen datuak (CI), CI eta harrapaketen batez besteko luzeraren datuak (ML), CI eta luzeraren araberako harrapaketen datuak (CAL), CI eta adinaren araberako harrapaketen datuak (CAA), CI eta adinaren araberako abundantzia-indizearen datuak (IAA) eta CAA eta adinaren araberako abundantzia-indizearen datuak (CAA+IAA). Epealdi-luzeraren araberako agertokiak: 12 (1978tik 1989ra), 22 (1978tik 1999ra), 32 (1978tik 2009ra) eta 41 (1978tik 2018ra). Egiazkoa agertokian selektibitatea urtero aldatzen zen eta, ondorioz, zenbait lerro marraztu dira (lerro bat urteko). Beste agertoki guztietan simulazioen selektibitatearen mediana marraztu zen, SRA ereduak aldaketarik gabeko urteko selektibitatea kalkulatzen baitu.

3.4 Eztabaida

Kapitulu honetan, SRA ereduaren errendimendua aztertu da, "auto-proba" simulazioaren bidez. Horretarako, mota eta epealdi-luzera ezberdinetako datu-multzoak, ustiatze-maila ezberdina jasan duten populazioak, hasierako harrapaketa-biomasa orekari buruzko hipotesi ezberdinak eta arrantza-jardueraren selektibitateari buruzko zehaztasun ezberdinak erabili dira. Espeziearen populazioari eta ustiatze-mailari loturiko parametroen kalkuluak zehatzagoak ziren, oro har, zenbat eta datu gehiago erabili. Horrez gain, erabilitako datuen epealdi-luzera zenbat eta luzeagoa izan, are eta zehatzagoak ziren kalkuluak.

Datuak ustiatze-egoera batetik abiatzen zirenean, beste inolako datu osagarririk gabe, harrapaketa guztien eta biomasa-indizearen datuen epealdiak espeziearen bi belaunaldi hartu behar zituen barnean gutxienez, kalkulu zehatzak eskuratu ahal izateko. Datu horien epealdia laburragoa zenean, simulazio gehienek "egiazko" baliotik urrun zeuden balioak eman zituzten. Ondorioz, kasu horietan, datu osagarriak erabili behar ziren kalkulu zehatzak lortzeko. Populazioa eta datuak arrantzarik gabeko egoera batetik abiatzen zirenean, SRA ereduaren kalkuluen zehaztasuna altua zen, epealdi laburrenaren kasuan ezik. Horrek, lotura zuzena izan dezake ustiatu gabeko egoera batetik hasten ziren populazioak sortzeko erabili den harrapaketen heriotza-tasaren ibilbidearekin. Izan ere, epealdi-luzera laburrenak, heriotza-tasaren ibilbidearen hasierako epealdia hartzen du, eta horren ondoriozko populazioaren murrizte-epea bakarrik hartzen du kontutan. Beste epealdi-luzera guztiek, ordea, murriztea eta populazioaren biomasaren egonkortzea hartzen dituzte barnean. Ono et al. (2014)k ikusi dutenez, epealdi-luzerak populazioaren murriztea eta populazioaren egonkortzea barnean hartzen dituenean, ebaluazio-ereduaren kalkuluen zehaztasuna handiagoa da, epealdi-luzerak populazioaren murrizte-epea soilik barnean hartzen duenean baino. Kapitulu honetan aztertutako epealdi guztiek, espeziearen murrizte-epealdia hartzen dute barnean, populazioaren egonkortze epealdia baina gehiago. Honela, eta Ono et al. (2014)ren arabera, ikerketa honetan aztertu diren epealdi-luzera guztiek informazio nahikoa eskaintzen duen harrapaketen heriotza-tasaren joera bat hartu dute aintzat. Konstanteagoa den harrapaketen heriotza-tasaren joera batek, kapitulu honetan lortutako zehaztasuna baino zehaztasun baxuagoko kalkuluak emango lituzke.

Eskuratu ziren kalkuluen zehaztasun altuak, lotura zuzena izan dezake erabilitako biologia-parametroen prezisioarekin, hauek populazioaren "egiazko" balioetara finkatu baitziren SRA eredua aplikatzeko. Biologia-parametro batzuk beste batzuk baina eragin handiagoa dute ebaluazio-ereduaren emaitzetan, zuzenki edo okerki zehaztuak izanez gero. Hordyk et al. (2019)ek frogatu du, heldutasun-parametroari loturiko zehaztugabetasunak ez duela garrantzi handirik beste parametro edo datu-iturri batzuei loturiko zehaztugabetasunarekin alderatuta. Halaz guztiz, ikusi zuten, baita ere, heriotza-tasa naturalaren alborapenak, harrapaketetan eta indizearen datuetan alborapen baliokidea izatea baina eragin handiagoa duela, epe luzerako errendimendua eta espeziearen biomasa kalkulatzeko garaian. Espeziearen biologia-parametroak ezagutzen ez direnean, ebaluazio-ereduak emango lituzkeen kalkuluen zehaztasuna kapitulu honetan lortu dena baino baxuagoa izango litzateke.

Espezieak ebaluatzeko ereduek, datu-multzo ezberdinak erabil ditzazkete espeziearen biomasa- eta ustiatze-mailak kalkulatzeko (Maunder and Punt, 2013). Datu-multzo bakoitza biltzeko kostua era esanguratsuan alda daiteke (Begg et al., 2005; Dennis et al., 2015). Datuak biltzeko programa xumeak eta merkeak nahikoak izan daitezke harrapaketen batez besteko luzeraren datu zehatzak biltzeko. Luzeraren araberako harrapaketen datuak biltzeko edo adinaren araberako harrapaketen datuak biltzeko ekimenak aldiz garestiagoak dira. Adinaren araberako datuak biltzeko kostua altuagoa da, bilketa egiteko behar diren pertsonen kopuruari eta adina zehazteko gaitasunari dagokionez, luzeraren araberako datuak biltzeko horiena baino. Horrez gain, arrantza-flota bati loturiko datuak lortzea, arrantza-flotarekiko inolako loturarik ez duten datuak lortzea baino merkeagoa da. Arrantza-flota bati loturiko datuen informazioa, baina, eskasegia izan daiteke populazio-abundantziari buruzko informazioa lortzeko eta arrantza-flotarekiko inolako loturatik ez duten datuek berebiziko garrantzia izan dezakete populazioaren abundantziaren kalkulu zehatzak lortzeko. Ikerketa honetan ikus daitekeen bezalaxe, arrantza-jardueraren selektibitatea flota-jarduerari loturiko datuetan oinarrituz kalkulatu zenean, arrantza-flotari loturiko informazioaz gain, arrantza-flotarekiko inolako loturarik ez zuten datuek garrantzi handia zuten selektibitatearen kalkulu zehatzak lortzeko. Flota-jarduerarekiko inolako loturarik ez duten datuak izateak, populazioaren dinamika hobeto ulertzen lagutzen du. Datu-mugatuak dituzten espezieei ikerketa denbora eta inbertsio txikiak esleitzen zaizkie normalean, datuak biltzeko eta aipatuak aztertzeko (Bentley, 2014). Horren ondorioz, datu-mugatuak dituzten espezieek ez dute normalean arrantza-flotarekiko loturarik gabeko daturik izaten. Ono et al. (2014)k frogatu duenez, ebaluazio-ereduetan datu osagarriak erabiltzea garrantzitsuagoa da bizi laburreko, azkar hazten diren espezieen kasuetan. Ikerketa honetan, bizi ertaineko eta hazkunde-maila ertaineko espeziea erabili zen (ICES, 2019a); ikerketa honetan ikusitakoa, datu mota bakoitzaren eta epealdi-luzera bakoitzaren garrantzia kalkulu zuzenak eskuratu ahal izateko, ondorioz, bestelakoa izan daiteke bizi-ziklo (biziraupen eta hazkunde-maila esaterako) ezberdinak dituzten espezieen kasuan.

SRA ereduak behar dituen gutxieneko datuei (hau da, harrapaketa guztien eta biomasa-indizearen datuak) harrapaketen batez besteko luzeraren datuak sartzeak, luzeraren araberako edo adinaren araberako datuak gehitzeak baino eragi gutxiago izan zuen kalkuluen zehaztasuna handitzeari dagokionean. Luzeraren edo adinaren araberako datuak gehitzeak, arrantzarik gabeko errekrutatzearen kalkuluaren zehaztasunean eragiten zuen beste kalkuluen zehaztasunean baina gehiago. Ikerketa honetan, adinaren araberako datuek informazio osagarri eskasa eman zuten, luzeraren araberako datuen aldean. Ono et al. (2014)k frogatu duenez, luzera-adina erlazioaren bariazio-koefizientea baxua bada, luzeraren araberako datuek urteetan zeharreko kohorteen jarraipena segitzea ahalbidetzen dute, adinaren araberako datuen beharrik gabe. Legatzaren hazkunde somatikoaren aldakortasuna nahiko baxua da (CV = 0,15) Then et al. (2015)ek egin zuen balioen gainbegiratze baten arabera. Horrek, adinaren araberako datuak erabilita eta luzeraren araberako datuak erabilita lortu zen kalkulu nagusien zehaztasunean ikusitako alde txikia azalduko luke. Oro har, adinaren araberako datuek hazkunde somatikoari, heriotza-tasa naturalari eta espezie-errekrutatzearen dinamikari buruzko informazio zehatzagoa ematen dute, luzeraren araberako datuek baino (Chen et al., 2003; Magnusson and Hilborn, 2007). Ikerketa honetan, ordea, biologia-parametro horiek akatsik gabe ezagutzen ziren. Egoera erreal batean, biologia-parametroak ez direnean zehaztasunez ezagutzen, adinaren araberako datuak erabiltzeak alde esanguratsuak ekar ditzazke luzeraren araberako datuen erabileraren aurrean, kalkuluen zehaztasunari dagokionez. Azkenik, gure emaitzek berretsi zuten kalkulu nagusien zehaztasuna handiagoa dela adinaren araberako harrapaketen datuak eta adinaren araberako abundantzia-indizearen datuak batera erabili zirenean, bakarka erabili zirenean baino, Chen et al. (2003)ek eta Ono et al. (2014)k aurretik iradoki zuten bezalaxe. Horrela gertatzen zen, bereziki, kalkulu zehatzak lortzeko selektibitatea arrantza-jarduerari loturiko datuak oinarri hartuta kalkulatu zen kasuetan.

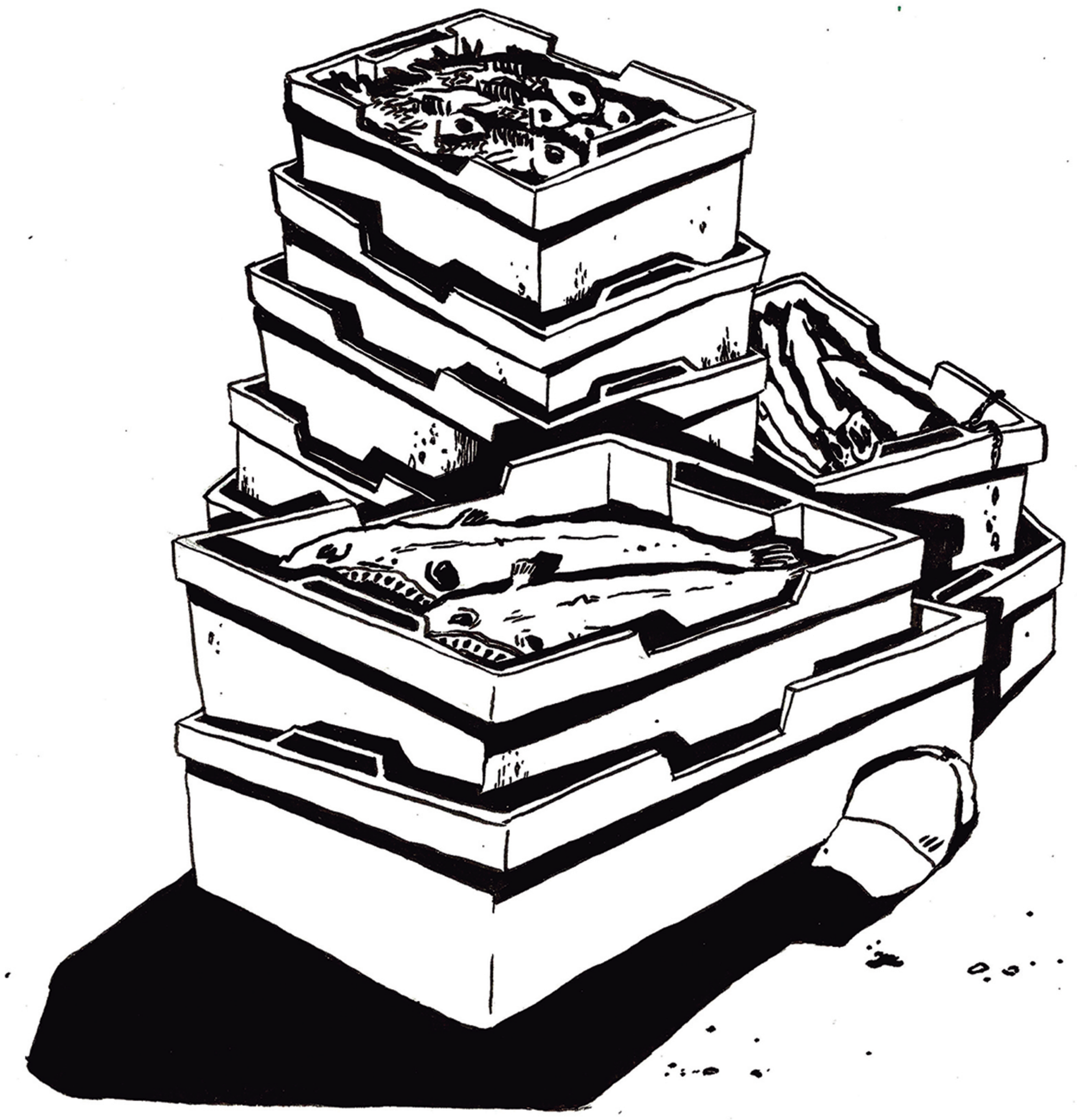
"Egiazko" arrantza-jardueraren selektibitate-patroiak aldakortasun handia agertzen du denboran zehar (ICES, 2019a), arrantza-flota gehienen kasuan gertatzen den bezalaxe (Sampson and Scott, 2012). Hala ere, ikerketa honetan ikusi genuen, denborarekin aldatzen ez den selektibitatea erabiliz lorturiko emaitzak, denborarekin aldatzen den selektibitatea erabiliz lorturikoak bezain zehatzak izan daitezkeela. Ikerketa honek nabarmendu du, baita ere, selektibitatea adinaren araberako harrapaketa datuetatik kalkulatu zen kasuetan, flota-jarduera bati loturiko adinaren araberako datuez gain, flota-jarduera batekiko inolako loturarik ez duten adinaren araberako datuak ere behar direla ebaluazio-eredu batetik kalkulu zehatzak lortzeko. Adinaren araberako abundantzia-indizearen daturik gabe, flota-jarduera bati loturiko adinaren edo luzeraren araberako datuetatik kalkulatutako selektibitateak, populazioaren abundantzia handiegiak kalkulatzen zituen. Legatzaren harrapaketetan, 7 urtetik beherako banakoak harrapatzen dira gehienbat. Plus adin-taldea, ordea, 15 urtetan finkatua dago legatzaren kasuan (ICES, 2019a). Selektibitateak populazioaren tamainaren datuak eta arrantza-flotaren harrapaketen datuak lotzen ditu. Adinaren araberako abundantzia-indizearen daturik gabe, eskuragarri dauden harrapaketen datuek ez dute banako zaharrenen abundantziari buruzko informaziorik eskaintzen. Honela, selektibitatea adinaren edo luzeraren araberako datuak erabiliz kalkulatu zenean, ereduak ziurtzat jo zuen arrantza-flotak harrapatu ez zituen banako zaharrenek heriotza-tasa naturala bakarrik pairatzen zutela eta, ondorioz, populazioaren abundantziaren kalkulua gehiegizkoa suertatu zen. Adinaren araberako abundantzia-indizearen daturik gabe, arrantza-flotak bere arreta banako txikietan jarrita badauka eta banako zaharrenen abundantzia baxua bada, emaitzek iradoki zuten, selektibitate-kurbaren beheranzko muga 0,25etik gorako edozein baliotan finkatu beharra zegoela ebaluazio-eredu batetik kalkulu zehatzak ateratzeko. Kasu honetan, selektibitate logistikoa zen selektibitate-hipotesirik egokiena kalkulu zehatzak lortzeko. Baina banako zaharrenak egiazki eskuragarri ez daudenean arrantza-flotarako, esaterako elasmobrankioen kasuan gertatzen den bezala (ICES, 2018d), selektibitate logistikoaren hipotesia desegokia izan daiteke eta selektibitatearen beheranzko muga balio baxuago batean zehazteak, selektibitate logistikoak baino kalkulu zehatzagoak eman ditzazke (ICES, 2018d). Honela, ganga itxurako selektibitatea ikus daiteke espezie batzuen kasuan, baina selektibitate logistikoa hipotesi zuhurrena da, ziurtzat jotzen baitu adin batzuetatik gora banako guztiak erabat zaurgarriak direla arrantza-jardueraren aurrean eta horrek biomasa-kalkulu baxuagoak eskaintzen baititu. Hordyk et al. (2019)ek arrain banako zaharrenen selektibitatearen alborapenari loturiko kudeaketa aholku okerren arriskua aztertu zuten. Agertoki guztietan selektibitate-kurbaren beheranzko muga balio ezberdinetan finkatu zuten, 0,25etik gora beti. Honela, frogatu zuten selektibitatearen zehaztapen ezak selektibitate-kurbaren beheranzko muga horretan ez duela apenas garrantzirik beste ziurgabetasun batzuekin alderatuta (adibidez, alborapena heriotza-tasa naturalean, makurduran edo harrapaketen edo indizearen datuetan).

Harrapaketa-biomasa orekari buruzko hipotesi ezberdinek emaitzetan zuten eragina altua izan zen guztizko harrapaketen datuak eta biomasa-indizearen datuak besterik erabili ez zirenean; baxua izan zen, aitzitik, datu osagarriak erabili zirenean, datu osagarri horiek harrapaketen hasierako heriotza-tasaren eta hasierako populazioaren inguruko informazioa eskaintzen baitzuten. Kalkulatutako arrantzarik gabeko errekrutatzearen, SSB aren eta murriztearen balioek gora egin zuten harrapaketa-biomasa orekari buruzko hipotesi altuagoa aplikatu zenean. Zenbat eta handiagoa izan zehaztutako harrapaketak oreka mailan, are eta biomasa handiagoa behar da harrapaketa horiek epe luzera sostengatzeko populazio-biomasan aldaketarik eman gabe (hau da, arrantzatu gabeko errekrutatze eta SSB altuagoaren beharra lehen urtean), eta are eta murrizte handiagoa simulazioaren lehen urtean. Oreka mailan zehaztutako harrapaketek, eragin txikiagoa izan zuten murriztearen kalkuluan, arrantzatu gabeko errekrutatze eta arrantzatu gabeko populazio-biomasa kalkuluetan baino. Harrapaketa-biomasa orekari buruzko hipotesiaren eragina txikia izan zen epealdiaren erdiko urteetako kalkuluetan. Harrapaketa-biomasa orekari buruzko hipotesiak, gehienbat, ereduaren lehen eta azken 7 urteei dagozkien kalkuluetan eragin zuen. Lehen eta azken urteetako kalkuluak gehi arrantzarik gabeko egoerako kalkuluak (adibidez, arrantzatu gabeko errekrutatzea) dira normalean kudeatzaileek kudeaketa neurriak ezartzeko erabiltzen dituzten parametroak; MSY bermatzen duen biomasa-maila, esaterako. R_0 kalkuluaren zehaztasunak menpekotasun handia agertzen zuen "egiazko" populazioaren hasierako murriztearekiko. Epealdi luzeak erabiltzeak, mesede egin zien, beste kalkulu nagusiei baino gehiago, azken urteko harrapaketen heriotza-tasaren eta azken urteko biomasa ugaltzailearen kalkuluei. Epealdi-luzera ezberdinetako agertoki guztietan, ordea, murriztearen kalkuluak beste kalkulu guztiak baino zehatzagoak ziren. Murriztea kalkulu erlatiboa da. Punt et al. (2002)k, Yin and Sampson (2004)k, Magnusson and Hilborn (2007)ek eta Deroba et al. (2014)k dagoeneko egiaztatu duten bezalaxe, zehaztasun handiagoz kalkulatzen dira populazioaren balore erlatiboak balore absolutuak baino. Ez denean "egiazko" populazioaren hasierako murriztemaila ezagutzen (eta, horrenbestez, ezta harrapaketa-biomasa oreka ere), emaitzek adierazten dute, kalkulu erlatiboetan (murriztea) oinarritutako kudeaketa plan bat, kalkulu absolutuetan (R_0) oinarritutako beste bat baino egokiagoa izan daitekeela. Kalkulu bakoitzaren zehaztasunaren eta alborapenaren inpaktua arrantzatzeko aukeren aholkuak emateko, baina, begizta itxiko simulazio-esparru batean aztertu behar da eta honek ikerketa osagarriak garatzea eskatzen du kudeaketari loturiko behin betiko aholkua eman baino lehen (Hordyk et al., 2019). Datuak ustiatze-egoera batetik abiatzen direnean, hasierako populazioaren hipotesi bat erabili ordez, epealdi-luzera berreraiki daiteke arrantzarik gabeko egoerara iritsi arte (Pauly and Zeller, 2016). Erakunde ezberdinak, ICES (ICES, 2021n), Sea Around Us (Pauly et al., 2020) eta FAO (FAO, 2020) esaterako, harrapaketen datu historikoak berreraikitzen eta biltzen saiatzen ari dira. Erronka handia da hori datozen urteetarako.

Ebaluazio-ereduetan parametro bakoitza ezagutzearen eta datu-iturri bat izatearen garrantzia kuantifikatzea garrantzitsua da populazio- eta ustiatze-mailen inguruko kalkuluen zehaztasuna hobetzeko erarik onena identifikatu ahal izateko. Horrek, bere aldetik, espeziearen kudeaketa hobetuko du. Harrapaketa-biomasa orekari buruzko hipotesiak kalkuluen zehaztasunean duen eragina ikusita (ikerketa honetan baieztatu den bezalaxe), simulazio osagarriak egin beharko lirateke. Simulazio horietan biologia-parametroen ezagutzan alborapena aplikatu beharko litzateke hasierako populazioaren inguruko hipotesi ezberdinekin batera, baita beste ebaluazio-hipotesi askorekin batera ere. Horrez gain, balore inizial batzuk biologia-parametroetan (makurduran edo heriotza-tasa naturalean, adibidez) eta eskala-parametroetan (esaterako, harrapagarritasun-koefizientean) erabiltzeak kalkuluetan duen eragina aztertu beharko litzateke. Etorkizuneko lan honek, parametro hauek zehaztasunez ezagutzen ez direnean SRA ereduak kalkulu zehatzak lortzeko duen gaitasuna aztertzen lagunduko luke.

Kapitulu honetan, datu-mugatuak dituzten espezieen populazio-dinamikaren kalkulu zehatzak lortzeko SRA ereduaren erabilera potentziala ebaluatu da, eta laburbilduz, ondokoak dira kapitulu honetatik eratorritako gomendioak:

orokorrean, orduan eta datu gehiago eta epealdi-luzeagoa erabili, orduan eta kalkulu zehatzagoak lortzen dira. Eskuragarri dauden datuak arrantzarik gabeko egoera batetik abiatzen direnean, espeziearen belaunaldi bateko luzera duten guztizko harrapaketa eta abundantzia-indizea erabiltzea nahikoa izango litzateke populazio- eta ustiatze-mailen inguruko kalkulu zehatzak lortzeko. Datuak ustiatze-egoera batetik hasten direnean aldiz, bi belaunaldi gutxienez barnean hartzen dituen guztizko harrapaketa eta abundantzia-indizea beharko lirateke ebaluazio-eredutik kalkulu zehatzak lortzeko. Epealdi horren luzera bi belaunalditik beherakoa bada, datu osagarriak beharko dira kalkulu zehatzak lortzeko. Espezie baten belaunaldi oso bat barnean hartzen duen harrapaketen batez besteko luzeraren datuak nahikoak izan daitezke kalkulu erlatibo zehatzak (murriztea, adibidez) lortzeko. Arrantzatu gabeko errekrutatzeari buruzko kalkulu zehatzak lortzeko, aitzitik, espeziearen belaunaldi bat aintzat hartzen duen adinaren edo luzeraren araberako datuak behar dira. Espeziearen hazkunde somatikoaren aldakortasuna baxua bada eta biologia-parametroak ezagutzen badira, luzeraren araberako datuen edo adinaren araberako datuen erabilerak antzeko zehaztasuneko kalkuluak emango ditu. Espeziearen belaunaldi bat aintzat hartzen duen adinaren araberako abundantzia-indizea gehitzea, luzeraren edo adinaren araberako datuekin batera, kalkulu zehatzak lortzeko aukerarik onena da, arrantza-jardueraren selektibitatea ezagutzen ez denean bereziki. Luzeraren edo adinaren araberako datuak ezezagunak direnean, kalkulu zehatzak lortzeko beste aukera bat da hasierako populazioari buruzko hipotesia hobetzea. Ezinezkoa denean denboran atzera eginaz ustiatze-epealdi historikoaren harrapaketen datuak berreraikitzea, harrapaketen hasierako heriotza-tasaren inguruko informazio osagarria beharko da populazioaren hasierako murriztearen eta, harrapaketa-biomasa orekari buruzko hipotesiaren gutxi gorabeherako ideia bat izateko. Hasierako populazioari buruzko hipotesi on bat funtsezkoa izango da ebaluazio-eredu batentzat kalkulu zehatzak lortzeko, gehienbat guztizko harrapaketen eta abundantzia-indizearen datuak baino ez ditugunean. Harrapaketa-biomasa orekari buruzko hipotesirik onena, lehen bost urteetako harrapaketen batez bestekoa izan daiteke. Hipotesi gisa hartutako oreka-mailan ziurgabetasuna gehitzea, ordea, mesedegarria izan daiteke ebaluazio-eredu batetik kalkulu zehatzagoak lortzeko, bereziki guztizko harrapaketen eta abundantzia-indizearen datuak baino ez ditugunean.



Chapter 4

Multiannual management strategy evaluation

4.1 Introduction

The demersal fishery that operates in the Bay of Biscay is a mixed-fishery, which implies that it catches several stocks simultaneously without the possibility of discriminating among them. The fishery caught more than 150 different species (Altuna-Etxabe et al., 2020; Briton et al., 2020). However, only the catches of approximately one-fourth of these species are regulated by TACs and quotas (Briton et al., 2020). Among the list of the 36 stocks or genus proposed by the European Commission to take into consideration in the multiannual plan to manage stocks and fisheries in the Western Waters and adjacent waters, only eight are caught by the demersal mixed-fisheries operating in the Bay of Biscay (EU, 2019b). These 36 species were selected because they are the most important target species considering all of the fisheries that operate in the European Western Waters and adjacent waters. However, most of the top-ranked species in Chapter 2 are not included. This raises the question on whether the current management measures are sufficient to ensure the sustainability of the system.

The scientific catch advice is usually given for each stock. However, in cases where the stocks have similar productivities and geographic distributions and/or similar morphological characteristics that made difficult to distinguish between them, the TACs and quotas are grouped and set at genus or even family level. The grouped TACs may provide benefits to fleets in terms of reducing the choking effect and offering flexibility to increase the quota of certain stocks. However, grouped TACs may lead to higher fishing mortality for each stock than recommended, potentially causing individual depletion. In the Bay of Biscay, grouped TACs are implemented for rays and anglerfishes. However, in the Bay of Biscay, their performance has never been evaluated and the existing evaluation of management strategies has been done at stock level (García et al., 2019).

Based on the results from the two previous chapters, this chapter focused on the development of the simulation model implemented by STECF in 2015 (STECF, 2015) to assess the impact of alternative management strategies on data-rich and data-limited stocks not previously considered. Specifically, a simulation algorithm was implemented to evaluate the impact of the current and alternative management strategies on the sustainability of the demersal mixed-fisheries operating in the Bay of Biscay. The analysis takes into account the implementation or non-implementation of the landing obligation, and aimed to provide a holistic evaluation of the management strategies. The real system was simulated including the dynamics of the demersal mixed-fisheries that operates in the area and 28 stocks identified with the species prioritisation approach developed in Chapter 2 as the most emblematic stocks. 13 stocks out of them were included with their age-structured population dynamics. The population dynamics of data-rich stocks were simulated based on their assessment model estimates, whereas the population dynamics of data-limited stocks, when it was possible according to the results obtained in Chapter 3, come from the SRA model fits. The rest 15 stocks were included assuming a constant CPUE approach. In addition, the factors that had the highest impact on the performance of the management strategy were identified. More precisely, the objectives of this chapter are 1) to evaluate the effect of alternative management strategies on the stocks sustainability when the HCR used was the same as the one used by ICES, 2) to test alternative HCRs for data-limited stocks, 3) to identify the factors that have the highest impact on the performance of the management strategy, and 4) to propose and test a management strategy that improves the performance of the system.

4.2 Materials and methods

The impact of various management strategies on the fishery and on the fish populations were analysed in a simulation framework. First, the fish and fleet dynamics were conditioned using historical data to determine the initial values and parameters necessary for future projections. Then, the simulation model took these initial values and projected the fishery and fish populations into the future while considering various management strategies. Finally, the performance of each management strategy was evaluated using a set of performance statistics. In addition, the factors that have the highest impact on the performance of the management strategy were investigated.

4.2.1 The initial fishery and fish populations

Overall, the exploitation of the fishery was categorized into 51 fleets (Table 4.1), while a total of 28 stocks (Table 4.2) were selected for inclusion in the simulation framework.

The fleets The demersal mixed-fishery in the Bay of Biscay is mainly formed by French and Spanish vessels (Figure F.18 in Appendix F). The fleets and metiers segmentation used was the one used in the provision of ICES mixed-fisheries considerations for the Bay of Biscay (ICES, 2022b,e). French and Spanish vessels were grouped in fleets that operated with the same gear, had the same vessel length, and belonged to the same country. The activity of the fleets was further divided into metiers that had the same group of target stocks and used the same mesh size. This resulted in 16 fleet and 39 metier combinations for French vessels, and 7 fleet and 12 metier combinations for Spanish vessels (Table 4.1).

Most of the stocks have a wider distribution than the area covered by the case study, and/or they are caught by other fleets in the Bay of Biscay. Hence, to account for the catches that occur outside the Bay of Biscay, an artificial fleet was created for each stock. These fleets had a single metier and resulted in 28 additional fleets (one per each stock). The catch data of the stocks at the metier level used to condition the fishery were obtained from the data used in the ICES mixed-fisheries working group (ICES, 2022b,e).

The stocks Considering all the stocks caught by a demersal mixed-fisheries in a simulation framework is practically impossible due to computational, data and time limitations. In this study, 28 stocks were included in the simulation model (Table 4.2). 25 stocks were selected based on the species prioritisation approach conducted in Chapter 2 for the Basque demersal mixed-fisheries. The selection was based on each stock's contribution to the total catch and income of the fishery, as well as their potential risk for the effects of fishing. The remaining 3 stocks (undulate ray, pollack and Norway lobster) were also included in the simulation because they are in the TAC and quota system, and the probability of their catches to limit the fishing activity of the demersal mixed-fisheries operating in the Bay of Biscay was high (ICES, 2022b,e). Spurdog (*Squalus acanthias*), ranked 19$^{\text{th}}$ in the species prioritisation (Chapter 2), was not included in the simulation due to its catch being forbidden (TAC equal to 0), so limits the fishing activity, and the demersal mixed-fisheries catch a low proportion of Spurdog (less than 3%) with respect to their overall catches (ICES, 2018e). These

Table 4.1: Fleets and metiers segmentation used in the simulation. Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). Target stocks: demersal fish (DEF), mix of fish (MIS), crustacean (CRU), mix of cephalopod and demersal fish (MCF) and mix of pelagic and demersal fish (MPD).

	Fleet			Metier		
	Gear	Vessels Length	Fleet name	Target	Mesh size	Metier name
French vessels	GNS	<10m	FR_GNS_<10m	DEF	100-119mm	GNS_DEF_100-119_0_0_all
					60-79mm	GNS_DEF_60-79_0_0
					all	GNS_DEF_all_0_0_all
		10<24m	FR_GNS_10<24m	DEF	100-119mm	GNS_DEF_100-119_0_0_all
					60-79mm	GNS_DEF_60-79_0_0
					all	GNS_DEF_all_0_0_all
	GTR	10-24m	FR_GTR_10-24m	DEF	100-119mm	GTR_DEF_100-119_0_0_all
					40-59mm	GTR_DEF_40-59_0_0
					all	GTR_DEF_all_0_0_all
	LHM	all	FR_LHM_all	DEF	all	LHM_DEF
	LLS	<24m	FR_LLS_<24m	DEF	all	LLS_DEF
		24<40m	FR_LLS_24<40m	DEF	all	LLS_DEF
	MIS	all	FR_MIS_all	MIS	all	FR_MIS
	OTB	<10m	FR_OTB_<10m	CRU		OTB_CRU
					>=70mm	OTB_CRU_>=70_0_0
					all	OTB_CRU_all_0_0_all
				DEF	<16mm	OTB_DEF_<16_0_0_all
					>=70mm	OTB_DEF_>=70_0_0
					16-31mm	OTB_DEF_16-31_0_0
					32-69mm	OTB_DEF_32-69_0_0
		10<24m	FR_OTB_10<24m	CRU		OTB_CRU
					>=70mm	OTB_CRU_>=70_0_0
					all	OTB_CRU_all_0_0_all
				DEF	<16mm	OTB_DEF_<16_0_0_all
					>=70mm	OTB_DEF_>=70_0_0
					16-31mm	OTB_DEF_16-31_0_0
					32-69mm	OTB_DEF_32-69_0_0
		24<40m	FR_OTB_24<40m	DEF	<16mm	OTB_DEF_<16_0_0_all
					>=70mm	OTB_DEF_>=70_0_0
	OTM	<10	FR_OTM_<10	DEF	all	OTM_DEF
		10<24m	FR_OTM_10<24m	DEF	all	OTM_DEF
					32-69mm	OTM_DEF_32-69_0_0_all
					70-99mm	OTM_DEF_70-99_0_0_all
		24<40m	FR_OTM_24<40m	DEF	all	OTM_DEF
					70-99mm	OTM_DEF_70-99_0_0_all
	OTT	10<24m	FR_OTT_10<24m	DEF	all	OTT_DEF
		24<40m	FR_OTT_24<40m	DEF	all	OTT_DEF
	SSC	10<40m	FR_SSC_10<40m	DEF	70-99mm	SSC_DEF_70-99_0_0_all
					all	SSC_DEF_all_0_0_all
Spanish vessels	GNS	10<24m	SP_GNS_10<24m	DEF	>=100mm	GNS_DEF_>=100_0_0
					60-79mm	GNS_DEF_60-79_0_0
		24<40m	SP_GNS_24<40m	DEF	all	GNS_DEF
	GTR	10<24m	SP_GTR_10<24m	MIS	all	SP_GTR
	LLS	10<40m	SP_LLS_10<40m	MIS	all	SP_LLS
	OTB	24<40m	SP_OTB_24<40m	DEF	all	OTB_DEF
				MCF	>=70mm	OTB_MCF_>=70_0_0
				MPD	all	OTB_MPD
		>=40m	SP_OTB_>=40m	DEF	all	OTB_DEF
				MCF	>=70mm	OTB_MCF_>=70_0_0
				MPD	all	OTB_MPD
	PTB	24<40m	SP_PTB_24<40m	MIS	all	SP_PTB

28 stocks represent more than 95% of the total catches and revenues of the Basque demersal mixed-fishery in the Bay of Biscay (Chapter 2). The stocks included in the simulation are ordered in Table 4.2 according to their position in the species ranking obtained in Chapter 2.

Hake, black anglerfish, white anglerfish, horse mackerel, mackerel, megrim, sea bass, blue whiting and sole are in ICES category 1 and have analytical assessment that provide abundance and exploitation level estimates (Table 4.3). These estimates were used to condition their population dynamics in FLBEIA (García et al., 2017). The summary of the assessment of these stocks is in Section F.1 in Appendix F.

Table 4.3: Stocks with an age-structured assessment approved by ICES. ICES nomenclature is used to define the stocks.

Species	**Stock code**	**Assessment model**	**Reference**
Hake	hke.27.3a46-8abd	Length-based Stock Synthesis model (SS3)	ICES (2021c)
Black anglerfish	ank.27.78abd	Length-based Stock Synthesis model (SS3)	ICES (2022c)
White anglerfish	mon.27.78abd	Age-based analytical assessment (a4a)	ICES (2021o)
Horse mackerel	hom.27.2a4a5b6a7a-ce-k8	Age- and length-based analytical assessment (SS3)	ICES (2021d)
Mackerel	mac.27.nea	Age-based analytical model (SAM)	ICES (2021f)
Megrim	meg.27.7b-k8abd	Bayesian statistical catch-at-age model	ICES (2021g)
Sea bass	bss.27.8ab	Age- and length-based analytical assessment (SS3)	ICES (2021k)
Blue whiting	whb.27.1-91214	Age-based analytical assessment (SAM)	ICES (2021b)
Sole	sol.27.8ab	Age-based analytical assessment (XSA)	ICES (2021m)

The rest of the stocks did not have an analytical assessment. Among these stocks, according to Chapter 3, the data available for catshark, red mullet, thornback ray and cuckoo ray were enough to obtain reliable population dynamics estimates from the SRA model; the catch and index data covered less than two generation times but more than one generation time of the stock and the catch-at-length data were available. The conditioning of catshark, red mullet, thornback ray and cuckoo ray population dynamics was performed using the SRA model available in the MSEtool package version 3.3.0 (Hordyk et al., 2021) within the R environment (R Core Team, 2022). The main differences between the SRA model available in MSEtool package version 3.3.0 (Hordyk et al., 2021) and the one available in the MSEtool package version 2.0.1 (Huynh et al., 2020a), which was tested in Chapter 3, is that the new version allows for the inclusion of uncertainty in total catch data and the assumption of fished equilibrium, whereas the old version assumed a fixed low error of 0.01. Thus, using the SRA model available in the new version, we estimated the population dynamics of catshark, red mullet, thornback ray and cuckoo ray, including uncertainty from all sources (such as, catch data, abundance index data, biological parameters, and initial population assumption). In the absence of knowledge about the uncertainty around the data and parameters used to condition their population dynamics, a uniform distribution around the mean parameter value was used in the SRA model to represent the uncertainty, considering that all the values within the range were equally plausible. In the most uncertain data and parameters (catch and biomass index data, natural mortality and recruitment parameters) a coefficient of variation of 10% was used whereas for more certain parameters (length-weight relationship, growth and maturity) a coefficient of variation of 5% was used. Uncertainty in fished equilibrium assumption was generated by multiplying the mean of the first five years

Table 4.2: Stocks included in the management strategy evaluation framework ordered according to their position in the species prioritisation ranking (Chapter 2). Stocks common and scientific name, distribution area using ICES nomenclature, position in the species prioritisation ranking (Rank), category in the ICES data availability classification (Cat), the population growth model used to include them in the simulation (age-structured population growth function, ASPG, or constant catch per unit effort, CPUE). Columns TAC and MAP are filled in with an "X" if the stocks are included in the TAC and quota system and/or in the Western Waters and adjacent waters multiannual management plan respectively. ICES ref indicates the main reference of ICES used for their conditioning in the simulation model.

Common name	Scientific name	Distribution area	Rank	Cat	Pop	TAC	MAP	ICES ref
Hake	*Merluccius merluccius*	Subareas 4, 6 and 7, and divisions 3a and 8abd	1	1	ASPG	X	X	ICES (2021c)
Black anglerfish	*Lophius budegassa*	Subarea 7, and divisions 8abd	2	1	ASPG	X	X	ICES (2022c)
Red mullet	*Mullus surmuletus*	Subareas 6 and 8, and divisions 7a–c, e–k and 9a	3	5	ASPG			ICES (2020b)
Catshark	*Scyliorhinus canicula*	Divisions 8abd	4	3	ASPG			ICES (2021e)
European squid	*Loligo vulgaris*	Not defined	5	Not	CPUE			
White anglerfish	*Lophius piscatorius*	Subarea 7, and divisions 8abd	6	1	ASPG	X	X	ICES (2021o)
Horse mackerel	*Trachurus trachurus*	Subarea 8, and divisions 2a, 4a, 5b, 6a and 7a–c, e–k	7	1	ASPG	X		ICES (2021d)
Mackerel	*Scomber scombrus*	Subareas 1-7 and 14, and divisions 8a–e and 9a	8	1	ASPG	X		ICES (2021f)
Megrim	*Lepidorhombus whiffiagonis*	Divisions 7b–k and 8abd	9	1	ASPG	X	X	ICES (2021g)
Pouting	*Trisopterus luscus*	Not defined	10	Not	CPUE			
Sea bass	*Dicentrarchus labrax*	Divisions 8ab	11	1	ASPG		X	ICES (2021k)
Thornback ray	*Raja clavata*	Subarea 8	11	3	ASPG	X		ICES (2020d)
Tope shark	*Galeorhinus galeus*	Subareas 1–10, 12 and 14	12	5	CPUE			ICES (2019b)
European conger	*Conger conger*	Not defined	13	Not	CPUE			
John dory	*Zeus faber*	Not defined	14	Not	CPUE			
Cuttlefish	*Sepia officinalis*	Not defined	15	Not	CPUE			
Argentine	*Argentina sphyraena*	Not defined	16	Not	CPUE			
Smooth-hound	*Mustelus spp.*	Subareas 1–10, 12 and 14	17-21	3	CPUE			ICES (2021l)
Tub gurnard	*Chelidonichthys lucerna*	Not defined	18	Not	CPUE			
Blue whiting	*Micromesistius poutassou*	Subareas 1–9, 12 and 14	20	1	ASPG	X		ICES (2021b)
Whiting	*Merlangius merlangus*	Subarea 8, and division 9a	22	5	CPUE	X	X	ICES (2021p)
Cuckoo ray	*Leucoraja naevus*	Subareas 6-7, and divisions 8abd	23	3	ASPG	X		ICES (2020a)
Poor cod	*Trisopterus minutus*	Not defined	24	Not	CPUE			
Grey gurnard	*Eutrigla gurnardus*	Not defined	25	Not	CPUE			
Sole	*Solea solea*	Divisions 8ab	25	1	ASPG	X	X	ICES (2021m)
Undulate ray	*Raja undulata*	Divisions 8ab	35	6	CPUE	X		ICES (2020e)
Pollack	*Pollachius pollachius*	Subarea 8, and division 9.a	41	5	CPUE	X		ICES (2021i)
Norway lobster	*Nephrops norvegicus*	Divisions 8ab	51	1	CPUE	X	X	ICES (2021h)

catches with a uniform error sampled with mean equal to 1 and a coefficient of variation equal to 0.25. A detailed description of the settings and data used to apply the SRA model to those four stocks can be found in Section F.1 in Appendix F.

Currently, there is no method available to assess the age-structured or biomass population dynamics of the rest of the stocks, and their dynamics were simulated based on the constant CPUE approach. Although Norway lobster has an analytical assessment, since 2016 its assessment uses an underwater TV survey to obtain an absolute value of its biomass (ICES, 2022d), and it is not translatable to a classical population dynamics model. Thus, their population dynamics could not be simulated in the model according to their assessment, and it was included in the simulation using the constant CPUE approach.

4.2.2 Simulation framework

The simulation was carried out using FLBEIA model (García et al., 2017), developed in R (R Core Team, 2022) using FLR libraries (Kell et al., 2007). The conceptual diagram of the MSE framework is in Figure A.6.

We considered that the population was perfectly observed with a two-year time lag between the observed population and the implementation of the advice. However, the catch advice derived from the HCR was not perfectly implemented in reality. The difference between the advice and the real catch arises naturally from the complexities of mixed-fisheries, where it is not possible to fully achieve all single-stock TACs simultaneously. Each component of the simulation model is described in detail below.

1000 sets of historical population dynamics were generated for each stock to parameterize the population dynamics of the stocks in the simulation framework. The historical fishing effort and stock populations spanned from 2010 to 2020, and the simulation from 2021 to 2040.

4.2.2.1 Operating model

The OM is the part of the simulation framework that simulated the “real system”, composed by stocks, fleets and their interaction.

In FLBEIA, the stocks can be simulated using either age-structured population dynamics or aggregated biomass. However, when the population dynamics is unknown, these stocks are included in FLBEIA assuming a constant CPUE approach. In this study, 13 of the 28 selected stocks were included with age-structured population dynamics: 9 stocks were simulated using age-structured analytical assessments and 4 stocks population dynamics were based on SRA model results. The other 15 stocks were included in the simulation using a fixed population using a constant CPUE approach (Table 4.2).

The stocks populations were projected forward by combining the exponential survival equation (Quinn and Deriso, 1999) and a stock-recruitment relationship. Monte Carlo simulation was used to introduce uncertainty in the simulation of the population dynamics of stocks in ICES category 1. The population of these stocks were simulated including uncertainty in natural mortality, weight-at-age, maturity-at-age and initial numbers-at-age data using a uniform distribution with mean equal to the assessment model result and a coefficient of variation of 0.25. Their stock-recruitment relationships were simulated using a segmented regression fitted to the last historical three years data (2018-2020). The stock-recruitment model was fitted for each iteration, and the stock-recruitment relationship stochasticity was introduced by multiplying, in each year and iteration, a lognormal random error to the point estimated with the stock-recruitment model. The lognormal distribution used had a mean equal to 1 and a coefficient of variation of 0.25. The life-history parameters of the stocks assessed by the SRA model were equal to the values obtained from the assessment model. In fact, the SRA tool itself provided a probability distribution of life-history parameters for the historical period. In addition, the stock-recruitment relationship of the stocks assessed by the SRA model were conditioned as in the assessment model, i.e., using the Beverton and Holt stock-recruitment relationship, and the parameter values calculated in the assessment model. The rest of the stocks without any population dynamics assessement were included in the simulation model using the constant CPUE approach, whereby it was assumed that their productivity was independent of the intensity of the fishing activity. The catch data of the two stocks of smooth-bounds (*Mustelus asterias* and *Mustelus mustelus*) are recorded at genus level (*Mustelus spp.*) so they were also included at genus level in the simulation model.

The relationship between catch and effort was described using the Cobb-Douglas production model at age level (Cobb and Douglas, 1928). Cobb-Douglas uses catchability, effort and biomass to calculate the corresponding catch. Historical catchability was calculated using historical biomass and effort data in the Cobb-Douglas function. The catchability was equal to the catch divided by the product of biomass and effort. In the projection, the catchability was assumed constant and equal to the average catchability from 2018 to 2020. However, the catchability varied by iteration because the historical biomass of each stock varied. The effort share among metiers is given as input parameter based on the average of the last historical three years data (2018-2020). As an exception, the fishing effort of the artificial fleets created to account for the catches that occur outside the Bay of Biscay was assumed constant (fixed effort) when the stocks caught are not in the TAC and quota system, whereas it fully complies with its quota share when the stocks caught are in the TAC and quota system. Fishery selectivity-at-age was implicitly included in catchability.

4.2.2.2 Observation model

No observation error was included in the simulation model.

4.2.2.3 Management procedure

In the management procedure, the management advice was produced based on the observed populations and prespecified HCRs. No error in the assessment model was assumed.

Management advice The management advice for stocks in ICES category 1 (hake, black anglerfish, white anglerfish, horse mackerel, mackerel, megrim, sea bass, blue whiting and sole) provided by ICES was based on the MSY advice HCR ($\mathrm{HCR_{MSY}}$). This advice rule uses fishing mortality and biomass reference points (i.e., $\mathrm{F_{MSY}}$, $\mathrm{B_{lim}}$ and MSY $\mathrm{B_{trigger}}$ reference points) to provide management advice (ICES, 2022a). $\mathrm{F_{MSY}}$ is the fishing mortality that maintains the population biomass of a stock with the maximum reproduction rate. $\mathrm{B_{lim}}$ is the stock size below which the reproductive capacity is significantly reduced. MSY $\mathrm{B_{trigger}}$ is the lower bound of SSB when a stock is fished at $\mathrm{F_{MSY}}$. The $\mathrm{HCR_{MSY}}$ consists in comparing the SSB at the beginning of the advisory year with MSY $\mathrm{B_{trigger}}$ reference point (Figure 4.1). If the SSB is at or above MSY $\mathrm{B_{trigger}}$, the advised fishing mortality is set equal to $\mathrm{F_{MSY}}$. While when the SSB is below MSY $B_{trigger}$, the advised fishing mortality decreases linearly to the origin of the biomass and fishing mortality equal to O. When the SSB is below B_{lim} zero catch can be advised based on precautionary considerations. In this study, when the SSB is below B_{lim} zero catch was advised.

ICES gives management advice of thornback ray and cuckoo ray using 2 over 3 HCR (HCR_{2over3}). In fact, although in 2021 ICES began developing alternative low data demand HCRs to give an advice of stocks in ICES category 3 based on the MSY proxies (ICES, 2022a), still nowadays, empirical HCRs that follows the precautionary approach such as HCR_{2over3} are the most used by ICES to provide advice for stocks in ICES category 3 (ICES, 2020c). The HCR_{2over3} adjusts the advised catch with the trend of an available stock abundance index. This HCR sets the catch advice by multiplying the most recent advised catch by the average of the last two years abundance index values divided by the average of the preceding three years abundance index values. The change in advised catches between years is limited to no more than 20% (ICES, 2022a).

Catshark and red mullet are not included in the TAC and quota system (EU, 2022), and they are not subject to any specific regulation in terms of catch limits.

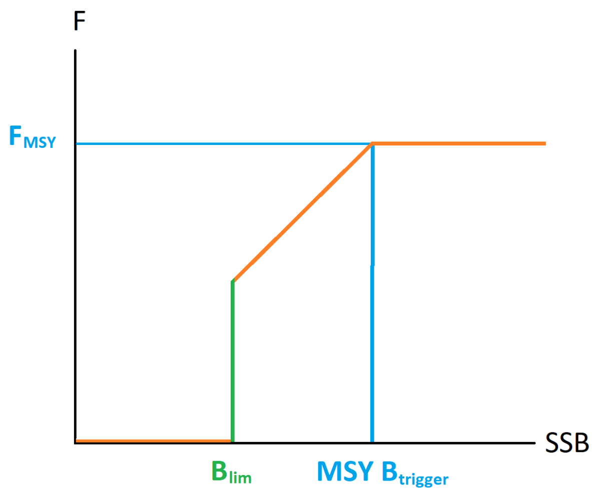


Figure 4.1: MSY advice rule used. The spawning stock biomass (SSB) at the beginning of the advisory year is compared with the MSY $B_{trigger}$ reference point. When the SSB is at or above MSY $B_{trigger}$, the advised fishing mortality is set equal to F_{MSY}. When the SSB is below MSY $B_{trigger}$ and above B_{lim}, the advised fishing mortality decreases linearly. When the SSB is below B_{lim} no fishing based on precautionary approach is advised.

For the rest of the stocks that are in ICES category 5 and 6 only catches or landings data time-series are available, and currently, there is not any HCR to generate the advice of these stocks. However, for whiting, undulate ray and pollack, a precautionary catch advice is given based on expert opinion. For those stocks and Norway lobster, a random variability was introduced in the projection of the TACs to consider possible changes in their TAC. The variability of the TACs was modeled as a uniform distributed with a mean equal to the average of the last three historical years' TACs and a coefficient of variation of 0.25.

4.2.2.4 Implementation model

The advice generated from the management procedure was implemented in the operating model, considering the dynamics of the fleets. In all the cases, the catches of all the stocks did not fully align with the advised catch due to the impossibility of fully achieving all single-stock TACs simultaneously. Consequently, the implementation error was indirectly included in the simulation through the fleet dynamics.

4.2.3 Scenarios

The base scenario was the fix scenario where the effort was constant along the time and equal to the average of the last historical three years effort data (2018-2020).

Two fleet dynamics scenarios were tested, motivated by the implementation or not of the landing obligation. In the first scenario, the fishery complied the landing obligation (min scenario). In this case, the fleets stop fishing once the TAC of one of the stocks caught by the fleets was reached. In the second scenario, the fishing effort was selected among the efforts corresponding to the stock quotas, choosing the effort that was most similar to the effort in the previous year (pre scenario).

For each fleet dynamics scenario three alternative management strategy scenarios were analysed using the HCRs that ICES uses to provide the fishing advice of each stock in the MSY framework and precautionary approach (i.e., HCR_{MSY} for stocks in ICES category 1 and HCR_{2over3} for stocks in ICES category 3). The first scenario mimicked the current management system (cu scenario). The stocks that are in the TAC and quota system (Table 4.2) were managed using TACs in a stock basis except for anglerfishes (black anglerfish and white anglerfish) and rays (thornback ray and cuckoo ray) that were managed at genus level. In the second scenario, all the stocks that are in the TAC and quota system were managed at stock basis (sepTAC scenario), and in the third scenario, only the stocks considered target stocks (i.e., those included in the multiannual management plan, see Table 4.2) were managed (tg scenario). The scenarios cu and tg differed for the stocks of horse mackerel, mackerel, blue whiting, thornback ray, cuckoo ray, undulate ray, pollack and sea bass (Table 4.2). The first seven stocks appear in the TAC and quota system but not in the multiannual management plan, so they were managed in cu scenario but not in tg scenario, and on the opposite, sea bass was in the multiannual management plan but not in the TAC and quota system, so it was managed in tg scenario but not in cu scenario.

Additionally, if adding new data-limited stocks in the TAC and quota system or if changing the HCR used to generate the advice of data-limited stocks (from HCR_{2over3} to HCR_{MSY}) improved the sustainability of the system was analysed. The biomass and fishing mortality reference points needed for the application of HCR_{MSY} are usually unknown for data-limited stocks. In this case, for data-limited stocks in HCR_{MSY} the 25% of the virgin SSB (B_0) was used as a proxy of B_{lim} (ICES, 2022g), and the MSY $B_{trigger}$ was defined as 140% of B_{lim} as it is the case for many stocks in ICES. The B_0 and fishing mortality reference point (F_{MSY}) values were taken from the SRA assessment model and varied by iteration. Apart from the default biomass and fishing mortality reference points, more precautionary reference points were also tested in the HCR_{MSY} application to improve the poor performance of the default biomass and fishing mortality reference point values in HCR_{MSY} for data-limited stocks. Two alternative scenarios to the default HCR_{MSY} application were tested. In one scenario, the default biomass reference points (MSY $B_{trigger}$ and B_{lim}) were increased by 50% (1.5B scenario). In the other scenario, the default fishing mortality reference point (F_{MSY}) was reduced by 50% (0.5F scenario). A diagram summarising all the scenarios settings is given in (Figure 4.2).

Finally, based on the results obtained from all the scenarios in Figure 4.2, we proposed a new scenario where each stock was managed by the best HCR identified (prop scenario). The proposed management strategy was evaluated with respect to the current management system (cu scenario) to see if it improved the sustainability of the system. The proposed management strategy scenario was also tested for implementation or not of the landing obligation (min and pre scenarios). Overall, 49 different scenarios were tested.

The default biomass (B_{lim} and MSY $B_{trigger}$) and fishing mortality (F_{MSY}) reference point values used in this study to apply the HCR_{MSY} are in Table 4.4.

Table 4.4: Biomass (in tonnes) and fishing mortality (year^{-1}) reference points for all the stocks included with a population dynamics. ICES nomenclature is used to define the stocks.

Species	**Stock code**	$\mathbf{B_{lim}}$	**MSY** $\mathbf{B_{trigger}}$	$\mathbf{F_{MSY}}$	**Reference**
Hake	hke.27.3a46-8abd	40,000	56,000	0.260	ICES (2021c)
Black anglerfish	ank.27.78abd	12,073	16,902	0.163	ICES (2022c)
White anglerfish	mon.27.78abd	16,032	22,445	0.280	ICES (2021o)
Horse mackerel	hom.27.2a4a5b6a7a-ce-k8	834,480	1,168,272	0.074	ICES (2021d)
Mackerel	mac.27.nea	2,000,000	2,800,000	0.260	ICES (2021f)
Megrim	meg.27.7b-k8abd	37,100	51,940	0.191	ICES (2021g)
Sea bass	bss.27.8ab	11,920	16,688	0.138	ICES (2021k)
Blue whiting	whb.27.1-91214	1,500,000	2,100,000	0.320	ICES (2021b)
Sole	sol.27.8ab	7,600	10,640	0.330	ICES (2021m)
Red mullet	mur.27.67a-ce-k89a	3,953	5,535	0.402	SRA assessment
Catshark	syc.27.8abd	7,445	10,424	0.295	SRA assessment
Thornback ray	rjc.27.8	1,888	2,644	0.113	SRA assessment
Cuckoo ray	rjn.27.678abd	25,356	35,499	0.742	SRA assessment

4.2.4 Performance statistics

The current and alternative management strategies were assessed in terms of three performance statistics:

Stock status The stock status in the projection period was evaluated based on the probability of the simulated SSB being below the median B_{lim}.

Fishing yield The effect that alternative management strategies had on fishing yield was evaluated by analysing the relative differences between the median catches of all the stocks over the projection years under alternative management strategy and the current management strategy (cu scenario).

Probability of being a choke stock Under the landing obligation, the probability of each stock being a choke stock was calculated by determining the probability of a stock to limit the fishing activity by fleet and year.

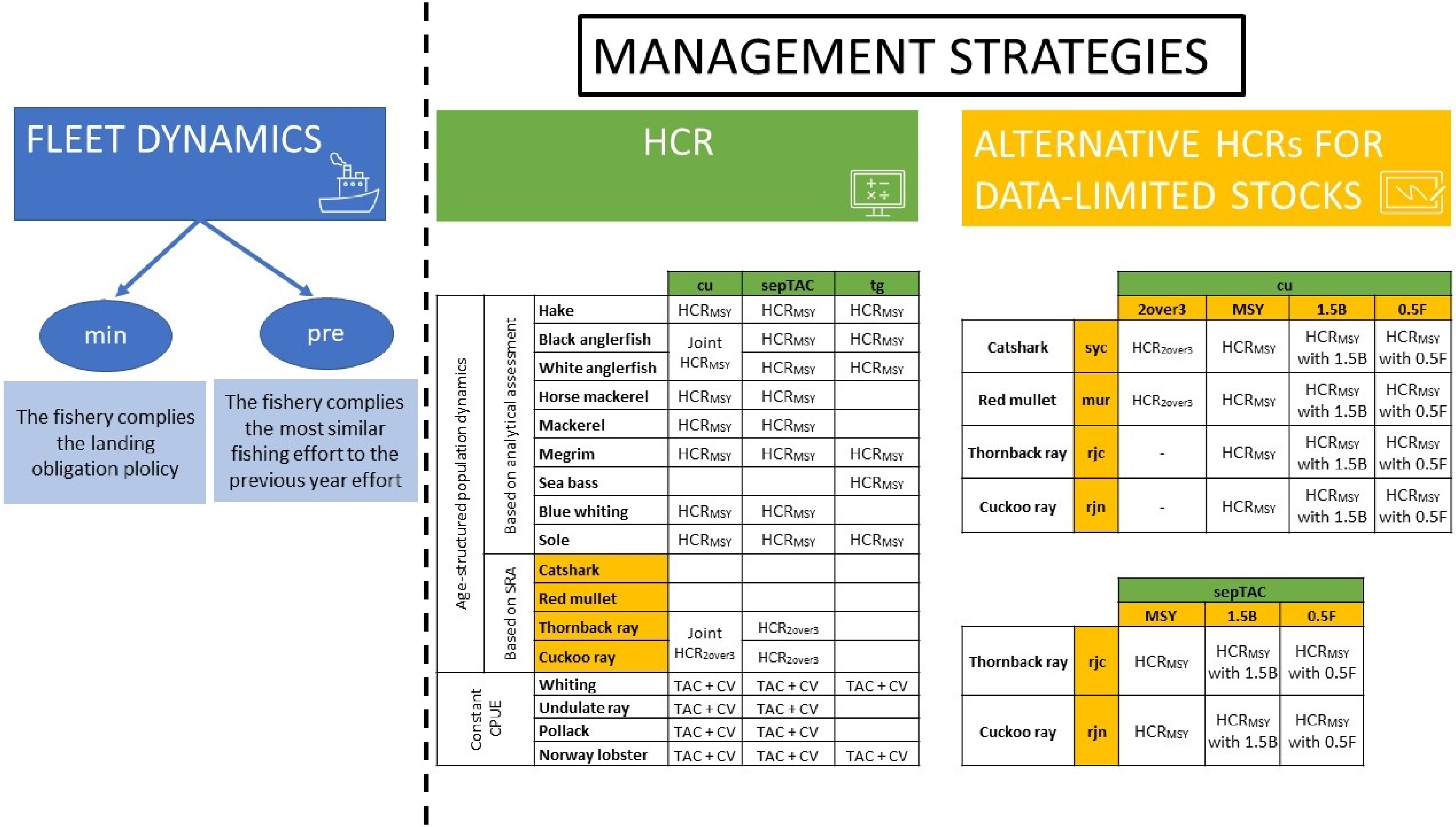


Figure 4.2: Scenarios tested in the simulation framework. For instance, "min_sepTAC" scenario was the scenario where the landing obligation was implemented, the stocks managed were those that are in the TAC and quota system and the advice of each stock was generated using the harvest control rule that ICES uses to provide management advice of them, and "pre_cu_mur_1.5B" scenario was the scenario where the landing obligation was not applied, the stocks managed were those that are in the TAC and quota system with the TACs of rays and anglerfishes grouped at genus level, and red mullet was included in the TAC and quota system giving its management advice using HCR_{MSY} with the biomass reference points equal to the double of default biomass reference points.

4.2.5 Identification of the factors that have the highest impact on the performance of the management strategy

For the data-limited stocks, the impact of changes in productivity parameters, starting condition (i.e., the SSB in the first year of the simulation) and management strategy attributes (i.e., implementation or not of the landing obligation, HCR used and grouped or separated TACs of rays and anglerfishes) was analysed.

As exploratory analysis, the relationship of the SSB in the last year of the simulation (2040) and the productivity parameters for each stock and each management strategy scenario was analysed. Four productivity parameters were used: the recruitment in unfished conditions (R_0), the virgin SSB (B_0), and the two stock-recruitment relationship parameters, α and β, where α represents the slope of the stock-recruitment relationship near the origin (biomass close to zero) and β represents the breaking point from which recruitment is considered independent of biomass. The correlation among productivity parameters, and between productivity parameters and the SSB in the starting year of the simulation (2021) was also analysed to identify the relationship among them.

A generalized linear model was applied individually for each stock to analyse the contribution of each explanatory variable on the response variable. The response variable was the SSB in the last year of the simulation (i.e., SSB in 2040), and the explanatory variables were the four productivity parameters, the SSB in the first year of the simulation (i.e., SSB in 2021), the implementation or not of the landing obligation, the HCR used and the grouped or separated TACs of rays and anglerfishes. Furthermore, the interaction between the stock-recruitment parameters and the landing obligation implementation, the interaction between the stock-recruitment parameters and the HCR applied, the iteration between the grouped or separated TACs and the HCR applied, and the iteration between the grouped or separated TACs and the landing obligation implementation were also included as explanatory variables.

4.3 Results

First, the effect of alternative management strategies on the sustainability of the stocks was tested using the same HCRs as those used by ICES in the MSY framework and precautionary approach. This involved applying the HCR_{MSY} for stocks in ICES category 1 and HCR_{2over3} for stocks in ICES category 3. Second, alternative HCRs were tested for data-limited stocks. Third, all of those management strategy scenarios were analysed to identify the factors that have the highest impact on the performance of the management strategy. Lastly, based on the results of all these scenarios, a management strategy that improves the overall performance of the system was proposed and tested.

4.3.1 Current harvest control rules

In this section, the impact of the current management strategy (cu scenario) in the SSB was assessed under two fleet dynamics scenarios (min and pre scenarios). In addition, the performance of the current management strategy was compared with the performance of the management strategy that included separate TACs for all the stocks in the TAC and quota system, and with the performance of the management strategy that includes only the management of target stocks. In all these scenarios, catch advice of the stocks was generated using the harvest control rule used by ICES in the MSY framework and precautionary approach (i.e., HCR_{MSY} for stocks in ICES category 1 and HCR_{2over3} for stocks in ICES category 3).

In all the scenarios, by the end of the simulation period, the median SSB of all the stocks was above the limit SSB (Figure 4.3). The limited impact of alternative management strategies in mackerel, horse mackerel and blue whiting (Figure 4.3), was due to the fact that a significant portion of these stocks' populations is distributed outside the Bay of Biscay, and most of the catch also occurs outside this area (Figure 4.3). For these stocks, the fix scenario was the only scenario that showed significant differences in comparison with the other scenarios (Figure 4.3). For the rest of the stocks, the median SSB varied among the scenarios. The differences in median SSB were bigger between scenarios where landing obligation was implemented or not (min and pre scenarios) than between alternative management strategy scenarios (cu, sepTAC and tg scenarios) (Figure 4.3). In the min scenario, which mimicked the implementation of the landing obligation, the SSB increased in most of the cases (Figure 4.3). However, in the pre scenario, where the landing obligation was not fully implemented, the SSB of most of the stocks remained stable or declined (Figure 4.3).

In the scenarios where the landing obligation is not implemented, the differences observed in the SSB for the different management strategy scenarios were small. This was true regardless of whether the TACs of rays and anglerfishes were grouped at genus level or not, and regardless of whether all the stocks in the TAC and quota system were managed or only target stocks were managed (Figure 4.3). However, in the scenario where the landing obligation was implemented, differences in SSB appeared between alternative management strategy scenarios. In this case, there were variations in the SSB depending on whether the TACs of rays and anglerfishes were aggregated at the genus level or not. The SSBs of catshark, cuckoo ray and particularly thornback ray were higher when separate TACs were used compared to when they were grouped (Figure 4.3). There were not significant differences in the SSB of anglerfishes when they were managed with separate or grouped TACs. Additionally, under the landing obligation, the long-term SSB of the stocks was higher when the current management strategy was used compared to when only some stocks in the TAC and quota system were managed (Figure 4.3). The biggest differences were observed for sole, sea bass, catshark, red mullet and rays (Figure 4.3).

In Figure 4.4 and Figure 4.5 the uncertainty bounds show the range of outcomes expected in 95% of the iterations due to uncertainty in recruitment and in other biological parameters. For data-limited stocks (catshark, red mullet, thornback ray and cuckoo ray) the SSB confidence intervals were very high (Figure 4.5). This happened due to the high uncertainty included to condition their population dynamics in order to account for plausible population dynamics parameters. The lower bound of the SSB was well above the limit SSB for all the stocks in ICES category 1 (Figure 4.4). However, for the data-limited stocks, the probability of SSB being below the limit SSB in the short- and long-term was higher than 10%, except for catshark that the probability was lower than 5% from 2025 onwards (Figure 4.5).

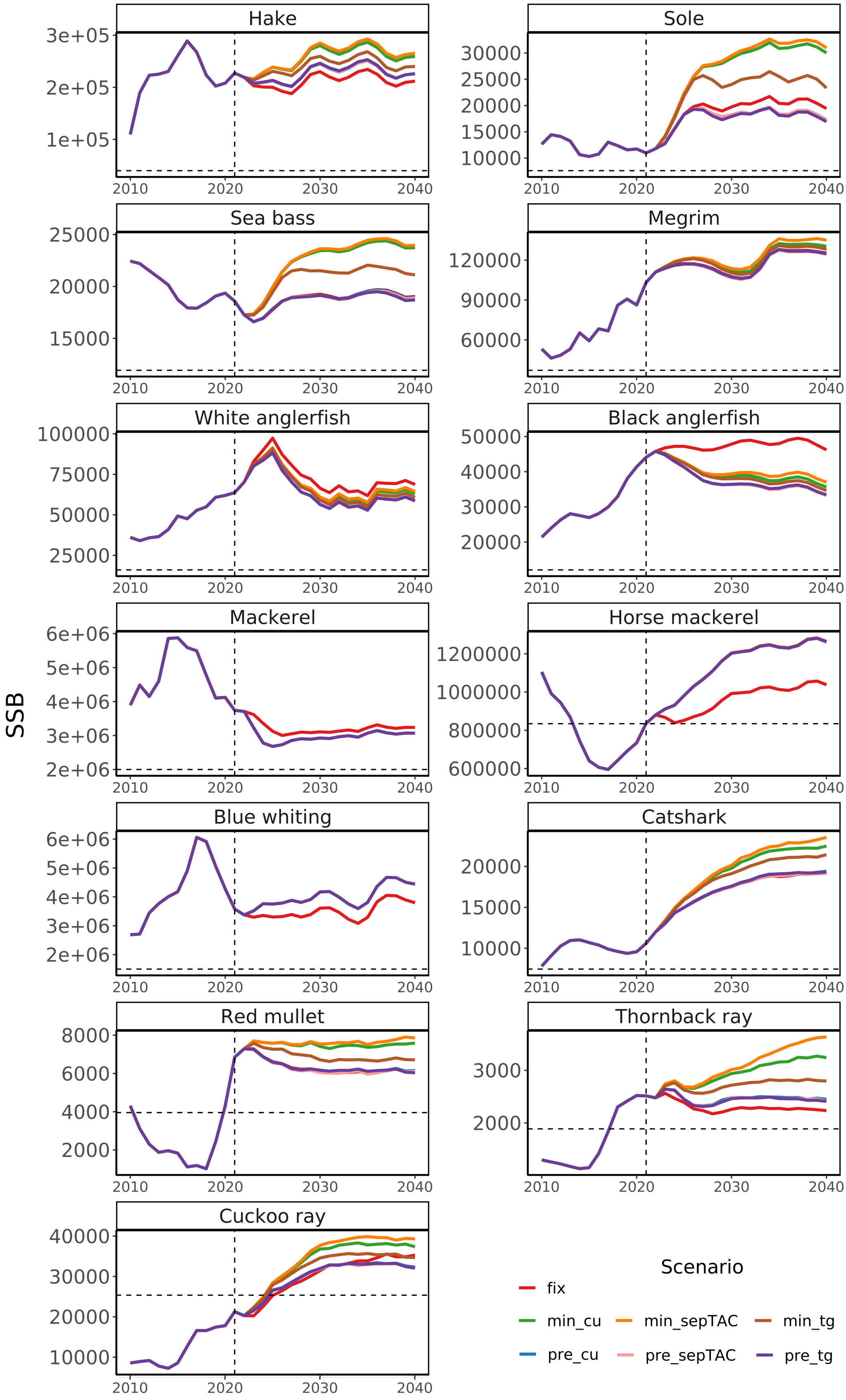


Figure 4.3: Median spawning stock biomass (SSB, in tonnes) for all the stocks with a dynamics population in the simulation when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. The starting year of the simulation (2021) and the limit SSB (B_{lim}) are shown as dashed black lines. The scenarios are defined in Figure 4.2

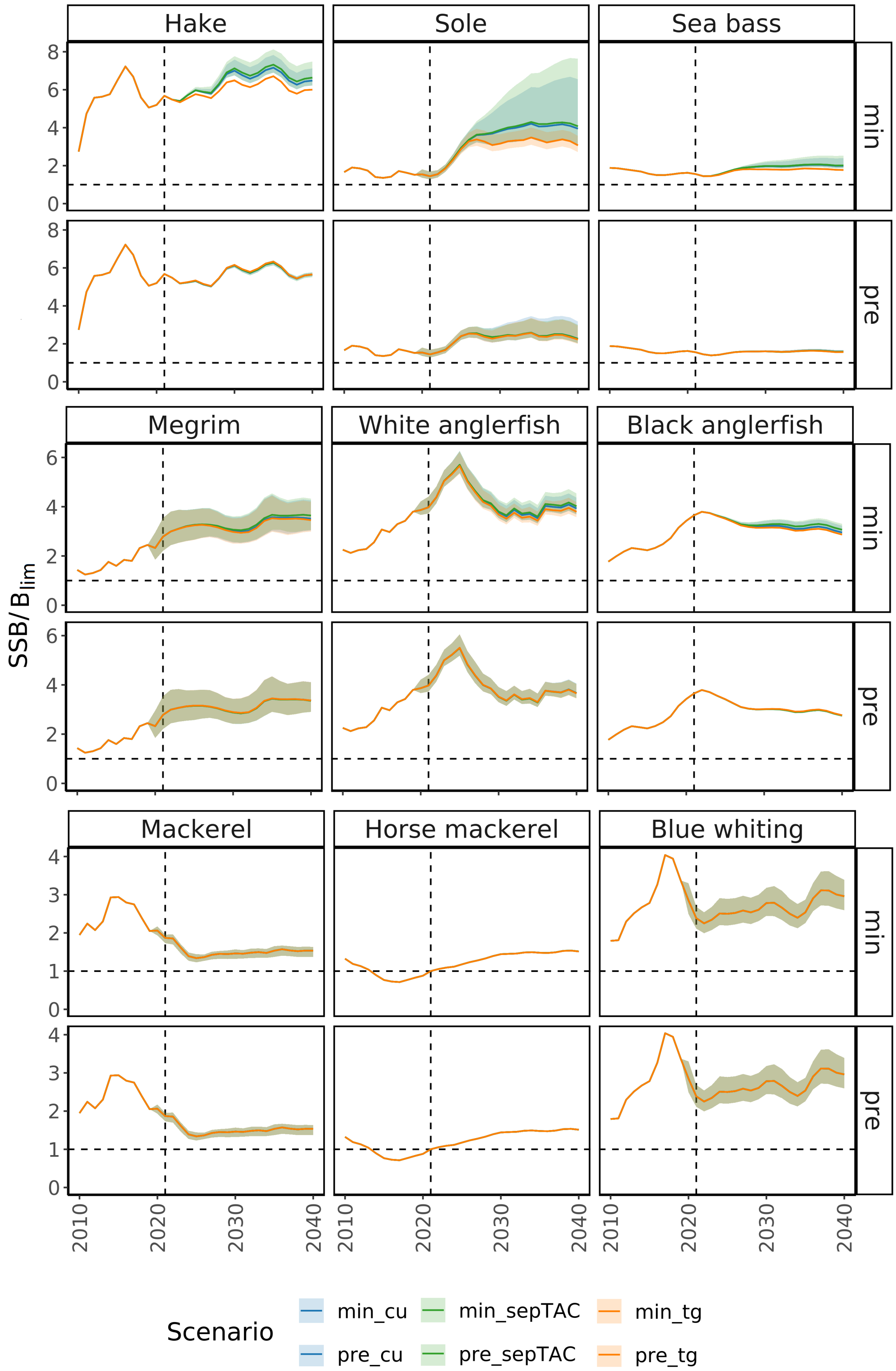


Figure 4.4: Trajectories of the spawning stock biomass (SSB) with respect to the limit SSB (B_{lim}) for data-rich stocks when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. Median values (colored lines) and variation (shaded areas; 5% and 95% quantiles) among iterations are shown. The starting year of the simulation (2021) and reference indicator value ($SSB/B_{lim} = 1.0$) are shown as dashed black lines. The scenarios are defined in Figure 4.2

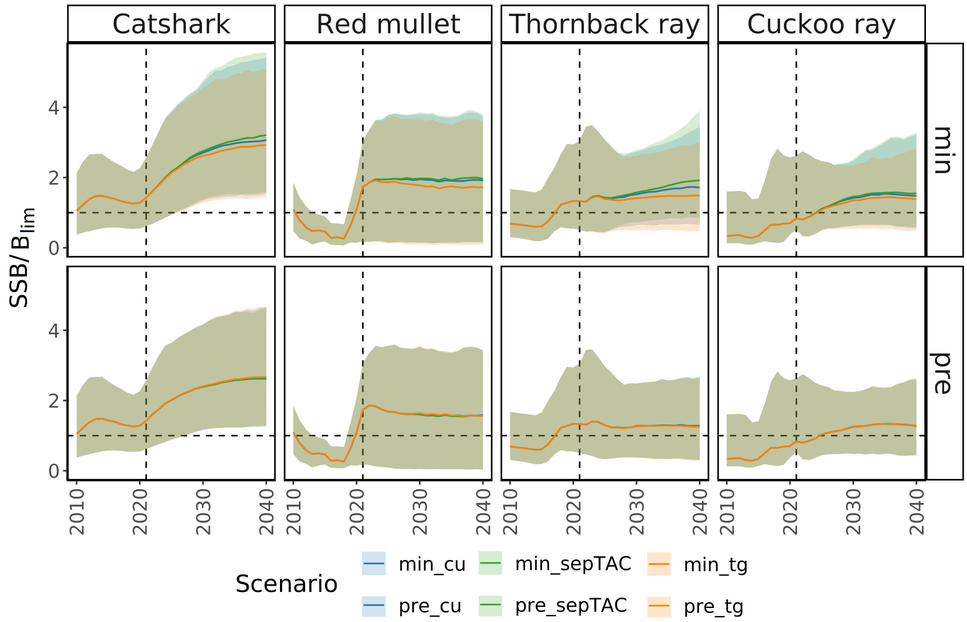


Figure 4.5: Trajectories of the spawning stock biomass (SSB) with respect to the limit SSB (B_{lim}) for data-limited stocks when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. Median values (colored lines) and variation (shaded areas; 5% and 95% quantiles) among iterations are shown. The starting year of the simulation (2021) and reference indicator value (SSB/B_{lim} = 1.0) are shown as dashed black lines. The scenarios are defined in Figure 4.2

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The probabilities of data-limited stocks' SSB being below the limit SSB in the first (2021) and last (2040) year of the simulation are given in Table 4.5.

Table 4.5: Probability of the spawning stock biomass (SSB) being below the limit SSB (B_{lim}) for data-limited stocks in the first (2021) and last (2040) year of the simulation when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. The scenarios are defined in Figure 4.2. The colors indicate the risk levels of the SSB being below B_{lim} (green for low, yellow for medium and red for high probability).

Scenario	Catshark (2021 = 0.22)	Red mullet (2021 = 0.17)	Thornback ray (2021 = 0.23)	Cuckoo ray (2021 = 0.55)
fix	0.01	0.37	0.41	0.39
min_cu	0.00	0.31	0.14	0.23
min_sepTAC	0.00	0.30	0.08	0.21
min_tg	0.00	0.35	0.24	0.28
pre_cu	0.01	0.37	0.33	0.34
pre_sepTAC	0.01	0.37	0.33	0.34
pre_tg	0.01	0.37	0.35	0.34

For catshark, the probability of SSB being below the limit SSB decreased in the simulation period to zero from 2029 onwards with the application of the landing obligation, and from 2035 onwards without it (Figure 4.6). However, the status of red mullet deteriorated along the simulation period (Figure 4.6). Both catshark and red mullet are not included in the TAC and quota system (Table 4.2). With the management of the stocks that are in the TAC and quota system or with the management of only the target stocks, the good state of the catshark population was ensured in the long-term. However, the good state of the red mullet population was not ensured. The probability of SSB being below B_{lim} was higher than 27% in all the management strategy scenarios and projection years.

Thornback ray and cuckoo ray are in the TAC and quota system, and currently, they are managed grouping their TACs at genus level. Thornback ray status deteriorated along the projection period when the landing obligation was not implemented (Figure 4.6). Under the implementation of the landing obligation, the stock status in the projection period remained almost constant when only the target stocks were considered. However, it improved by more than 10% when all the stocks in the TAC and quota system were used to constrain the fishing activity (Figure 4.6). In all the scenarios, the status of cuckoo ray improved in the projection period (Figure 4.6). For both rays, the HCR used to provide management advice for them (HCR_{2over3}) was not able to ensure their good state in the long-term (Figure 4.6).

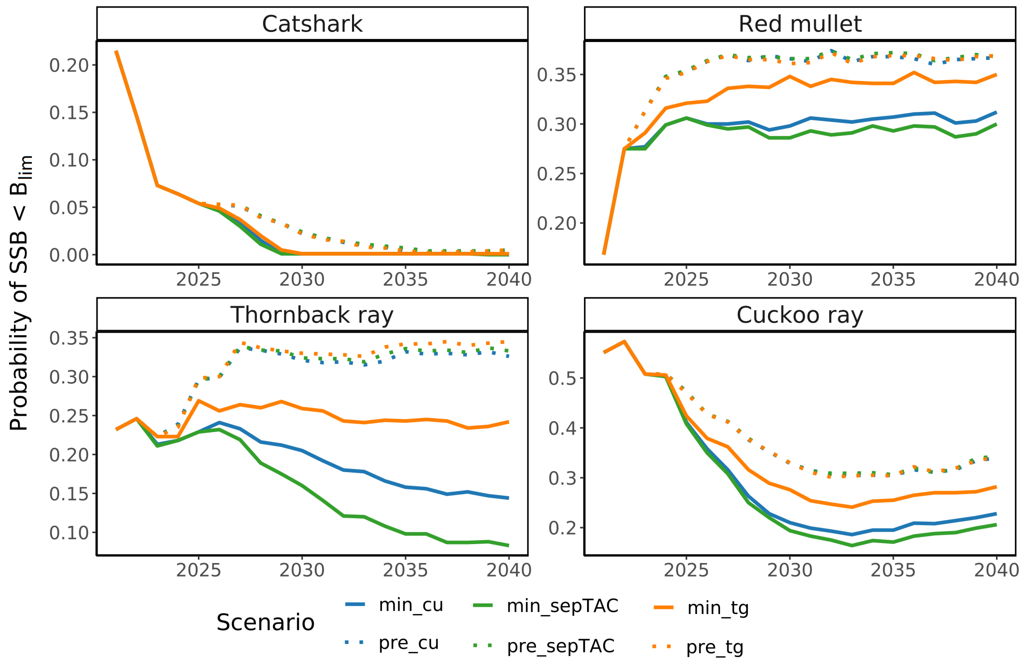

Figure 4.6: Probability of the spawning stock biomass (SSB) being below the limit SSB (B_{lim}) for data-limited stocks when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. The scenarios are defined in Figure 4.2

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Under the implementation of the landing obligation, the catch production of most of the fleets increased when only target stocks limited the fishing activity (Figure 4.7). However, when all the stocks in the TAC and quota system were used to limit the fishing activity and the TACs of rays and anglerfishes were used at stock level, the catch production decreased in most of the fleets with respect to the current management strategy (Figure 4.7). The same happened when the landing obligation was not implemented. Although in this case, the differences were only found in few fleets (Figure 4.7).

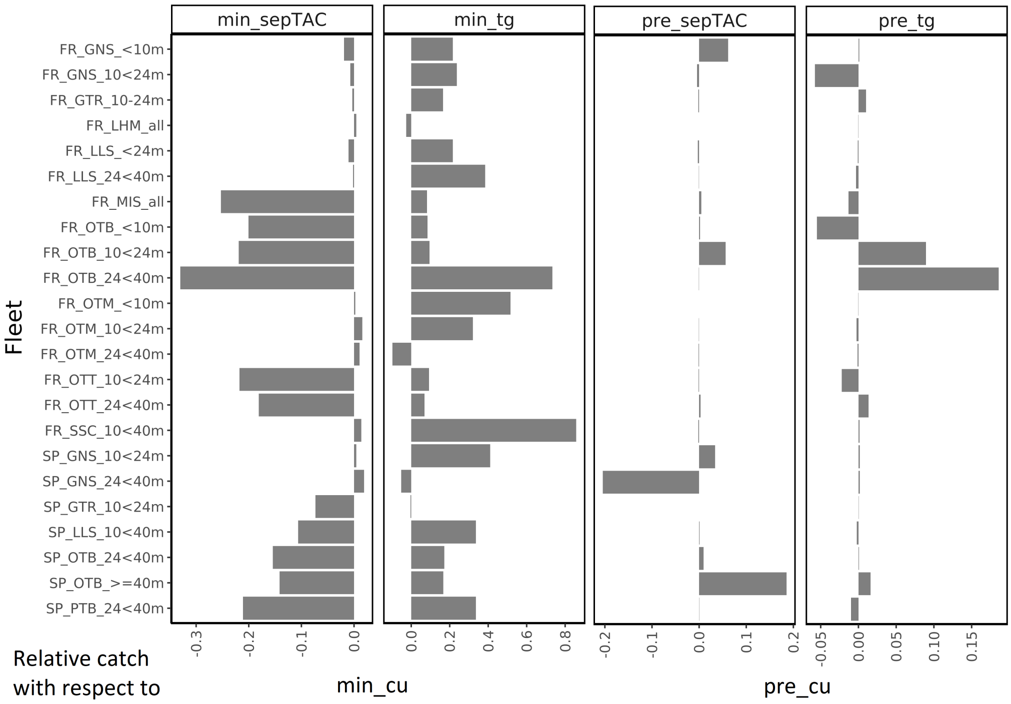


Figure 4.7: Relative difference between the catches of all the stocks over the projection years under alternative management strategies with respect to the currently implemented, when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. Country acronyms: French vessel (FR) and Spanish vessel (SP). Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). The numbers in fleet's name indicates the length of the vessels. The fleets and management strategy scenarios are defined in Table 4.1 and Figure 4.2 respectively.

Under the landing obligation framework, the probability of a given stock choking the effort of each fleet by year was evaluated considering all the stocks in the TAC and quota system. The results for the probabilities with grouped TACs for rays and anglerfishes are shown in Figure 4.8, while the probabilities with separated TACs are depicted in Figure 4.9. The probability of choking for each stock varied smoothly along time (Figure 4.8, Figure 4.9). Under the current management strategy, the probability of pollack being the one that limited the fishing activity was higher than 20% for most of the fleets, being 100% for French seiners, and higher than 70% for Spanish gillnetters with vessel length lower than 24m (Figure 4.8). The quota of rays was the most restrictive one for French gillnetters, trammel netters and longliners (Figure 4.8). The choking probability of sole was high for French handliners, mid water otter trawlers with vessel length higher than 24m and twin-rigged otter trawlers, and for Spanish gillnetters with vessel length higher than 24m and trammel netters (Figure 4.8), whereas the horse mackerel quota was the one that had the highest probability to constrain the fishing activity of French mix and bottom otter trawlers fleets, and Spanish longliners and bottom pair trawlers (Figure 4.8). The quota of whiting in Spanish bottom otter trawlers, the quota of mackerel in French mid water otter trawlers with vessel length lower than 10m, and the quota of blue whiting in French mid water otter trawlers with vessel length higher than 10m also had a high probability to limit the fishing activity (Figure 4.8).

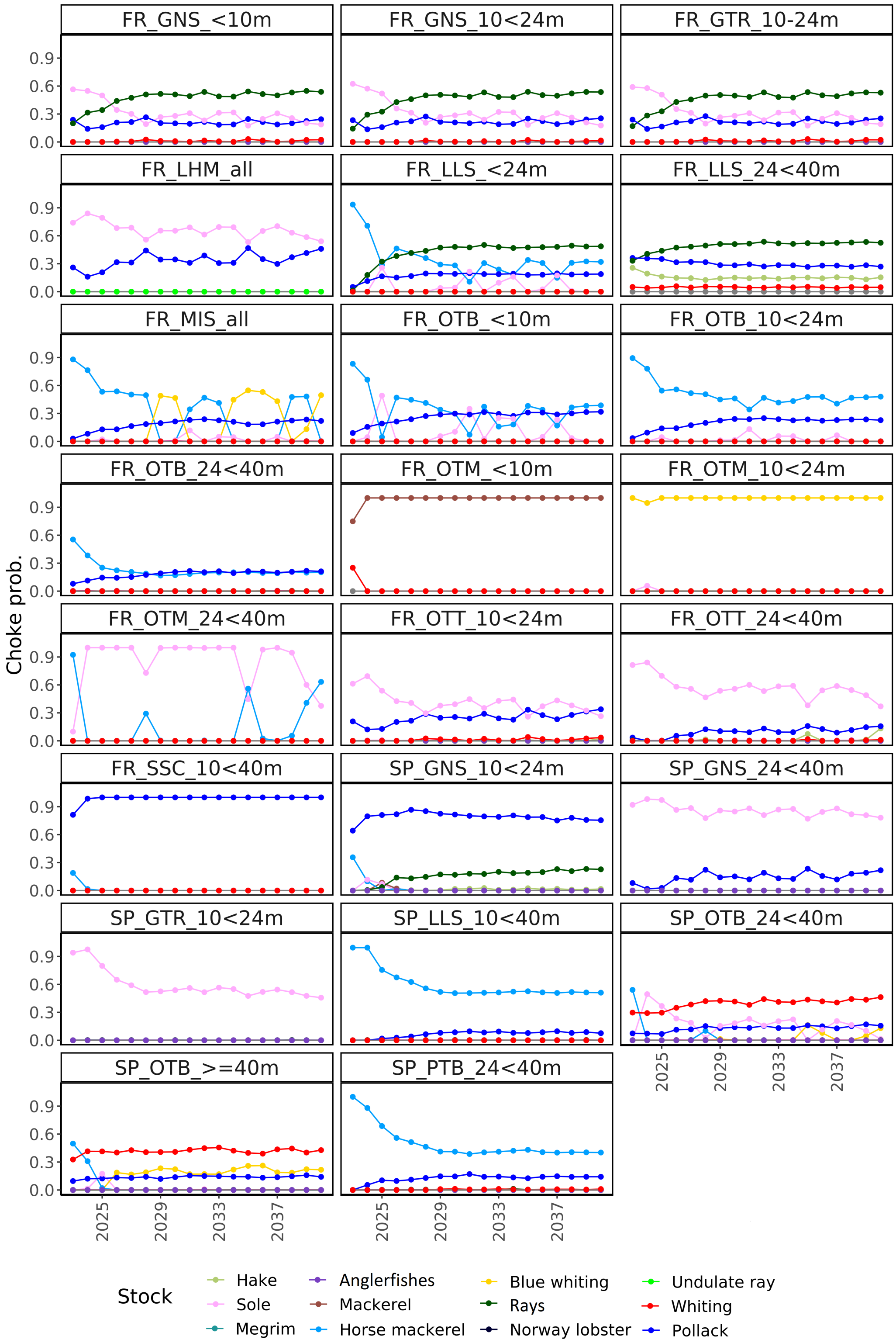


Figure 4.8: For each fleet, the probability of the effort being limited by each stock in each year of the simulation considering the current management strategy and under the landing obligation when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. Country acronyms: French vessel (FR) and Spanish vessel (SP). Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). The numbers in fleet's name indicates the length of the vessels. The fleets are defined in Table 4.1.

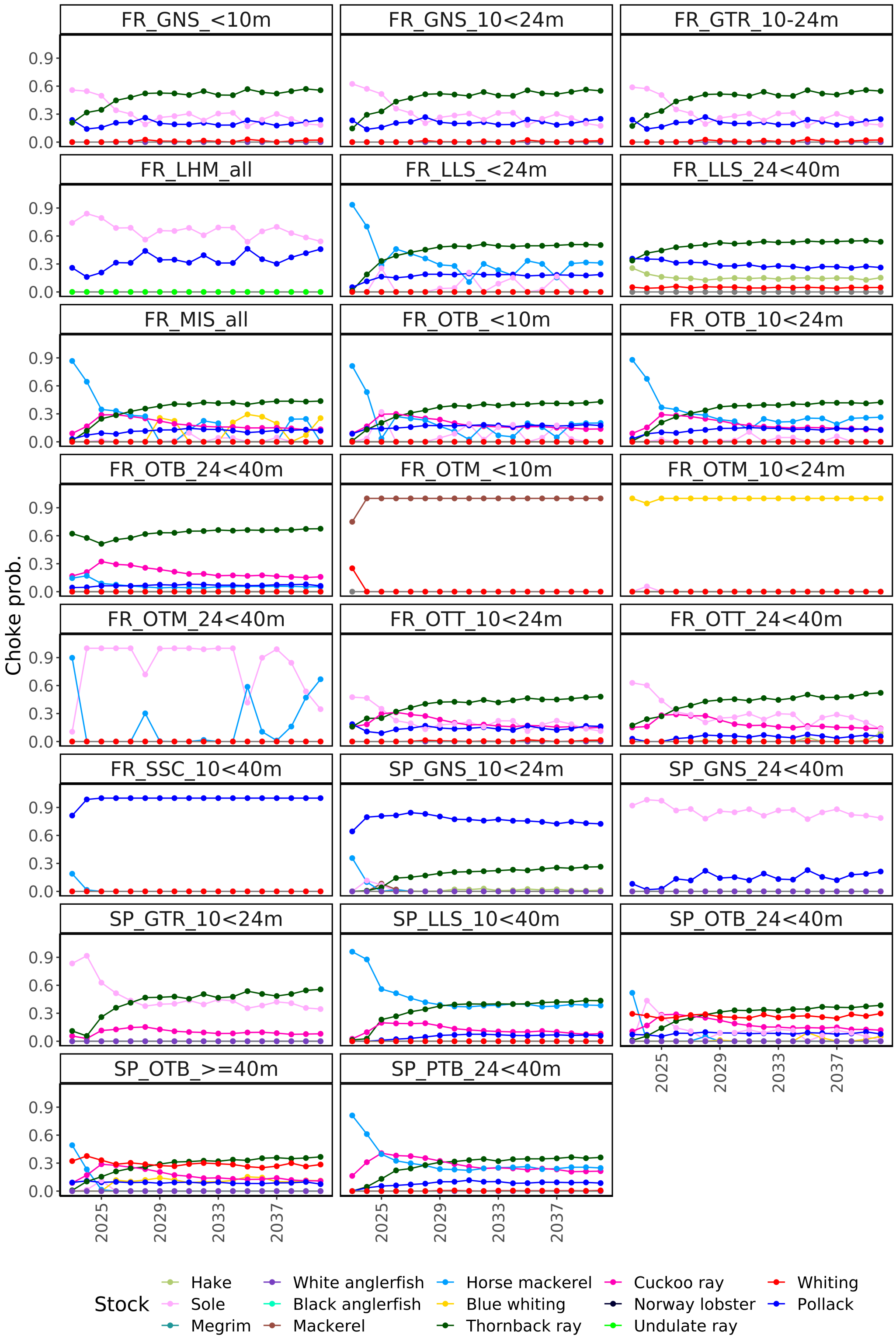


Figure 4.9: For each fleet, the probability of the effort being limited by each stock in each year of the simulation considering the TACs of all the stocks included in the TAC and quota system individually and under the landing obligation when the HCRs used were the same as the one used by ICES in the MSY framework and precautionary approach. Country acronyms: French vessel (FR) and Spanish vessel (SP). Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). The numbers in fleet's name indicates the length of the vessels. The fleets are defined in Table 4.1.

When the TACs of rays and anglerfishes were applied at stock level, the quota of thornback ray and cuckoo ray not only limited the fishing activity of French gillnetters, trammel netters and longliners, but also limited the French mixed fleet and twin-rigged otter trawlers, as well as Spanish trammel netters, longliners and bottom pair trawlers, along with French and Spanish bottom otter trawlers fishing activity (Figure 4.9). This means that when the TACs of rays was provided at genus level, the catch quota of both rays together helped to alleviate the choking effect of their individual quota.

4.3.2 Alternative harvest control rules for data-limited stocks

In this section, alternative HCRs to provide management advice of each of the data-limited stocks were investigated with the objective of identifying the HCRs that improved the sustainability of each stock with respect to the current management strategy. As alternative HCRs for each data-limited stock had marginal impact on other stocks, in this section, we specifically focused on the effect of alternative HCRs on the corresponding data-limited stock, namely catshark, red mullet, thornback ray and cuckoo ray.

The Table 4.6 presents the probability of the last year's SSB (2040) being below the limit SSB for the data-limited stocks under alternative HCR scenarios.

Table 4.6: Probability of the last year (2040) spawning stock biomass (SSB) being below the limit SSB (B_{lim}) for data-limited stocks under alternative HCR scenarios. The scenarios are defined in Figure 4.2. The colors indicate the risk levels of the SSB being below B_{lim} (green for low, yellow for medium and red for high probability).

Scenario	Catshark
min_cu_syc_2over3	0.00
min_cu_syc_MSY	0.00
min_cu_syc_1.5B	0.00
min_cu_syc_0.5F	0.00
pre_cu_syc_2over3	0.01
pre_cu_syc_MSY	0.01
pre_cu_syc_1.5B	0.01
pre_cu_syc_0.5F	0.01

Scenario	Red mullet
min_cu_mur_2over3	0.18
min_cu_mur_MSY	0.17
min_cu_mur_1.5B	0.08
min_cu_mur_0.5F	0.03
pre_cu_mur_2over3	0.34
pre_cu_mur_MSY	0.42
pre_cu_mur_1.5B	0.22
pre_cu_mur_0.5F	0.21

Scenario	Thornback ray
min_cu_rjc_MSY	0.09
min_cu_rjc_1.5B	0.02
min_cu_rjc_0.5F	0.02
min_sepTAC_rjc_MSY	0.06
min_sepTAC_rjc_1.5B	0.01
min_sepTAC_rjc_0.5F	0.01
pre_cu_rjc_MSY	0.27
pre_cu_rjc_1.5B	0.15
pre_cu_rjc_0.5F	0.16
pre_sepTAC_rjc_MSY	0.27
pre_sepTAC_rjc_1.5B	0.14
pre_sepTAC_rjc_0.5F	0.15

Scenario	Cuckoo ray
min_cu_rjn_MSY	0.71
min_cu_rjn_1.5B	0.46
min_cu_rjn_0.5F	0.61
min_sepTAC_rjn_MSY	0.33
min_sepTAC_rjn_1.5B	0.16
min_sepTAC_rjn_0.5F	0.24
pre_cu_rjn_MSY	0.72
pre_cu_rjn_1.5B	0.46
pre_cu_rjn_0.5F	0.63
pre_sepTAC_rjn_MSY	0.72
pre_sepTAC_rjn_1.5B	0.41
pre_sepTAC_rjn_0.5F	0.57

4.3.2.1 Catshark

When the fleets activity was constrained by the quota of catshark, in addition to the stocks that are in the TAC and quota system, the probability of SSB being below B_{lim} slightly decreased in a short-term when the landing obligation was implemented. With HCR_{2over3}, the performance improved with respect to the current management strategy, but less than when HCR_{MSY} was used (Figure 4.10). Without the landing obligation, alternative HCRs for managing catshark and the current management strategy did not show big differences (Figure 4.10). As for catshark, the SSB of most of the data-rich stocks was higher in the last year of the simulation than in the first year of the simulation (Figure 4.3). As a result, the overall system improved in the long-term compared to the short-term.

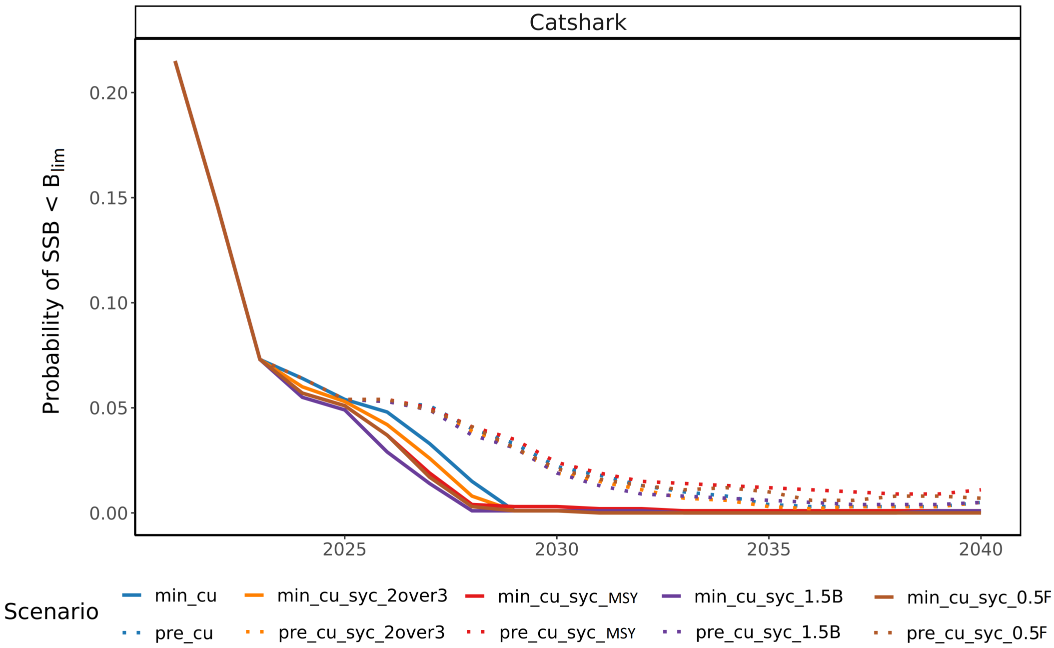


Figure 4.10: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for catshark (syc) management scenarios. The scenarios are defined in Figure 4.2.

4.3.2.2 Red mullet

Under the landing obligation, when red mullet was included in the TAC and quota system, HCR_{2over3} and HCR_{MSY} gave similar results, and the probability of the stock SSB falling below safe biological limits decreased by 15% with respect to the current management strategy (Figure 4.11). In addition, when the default fishing mortality reference point was reduced or when the default biomass reference points were increased, the probability of SSB being below the limit reference biomass decreased in an additional 15% (Figure 4.11). The lowest probability of SSB falling below B_{lim}, close to zero from 2027 onwards, was obtained in the scenario where the default fishing mortality reference point was reduced (Figure 4.11).

When the landing obligation was not implemented, the inclusion of red mullet in the TAC and quota system led to a decrease in the probability of SSB being below B_{lim} only when the TAC was generated using the HCR_{2over3} (Figure 4.11). In fact, the use of the HCR_{MSY} with default reference points resulted in higher catch and a higher probability of the stock SSB falling below safe biological limit than when it was not explicitly managed (Figure 4.11). In fact, the biomass and fishing mortality reference points used to generate the advice from HCR_{MSY} were too low and too high respectively (Table 4.4) that the TAC advice given by the HCR_{MSY} was higher than that given when it was not managed. When the reference points in the HCR_{MSY} application were more precautionary, the stock status in the long-term was better with HCR_{MSY} than with the current management strategy and with the scenario where HCR_{2over3} was used. However, none of the HCRs tested achieved full protection of the stock when the landing obligation was not implemented (Figure 4.11).

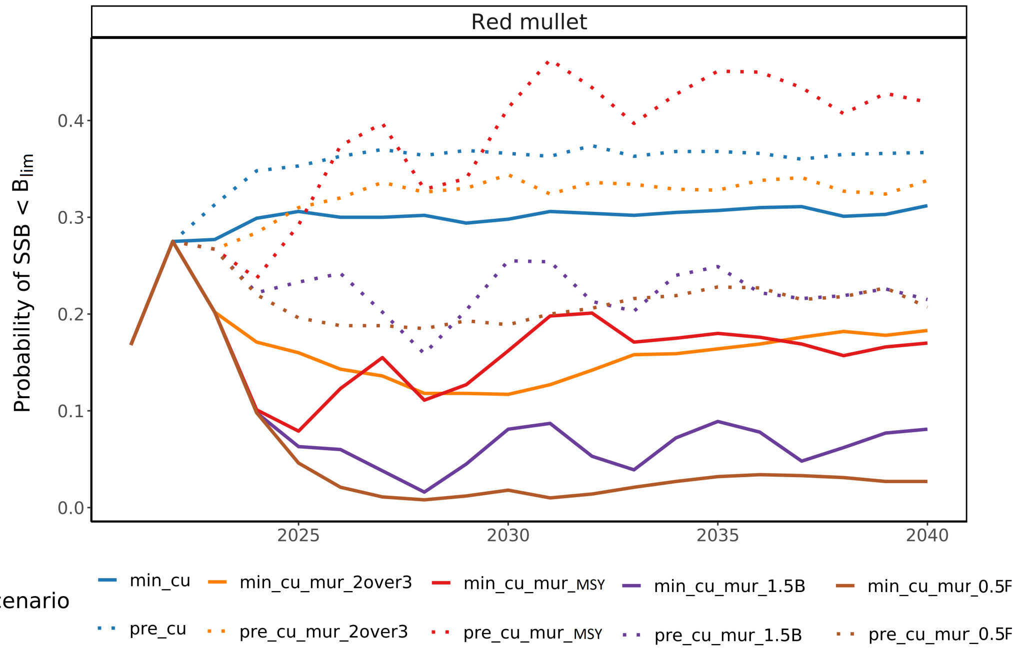


Figure 4.11: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for red mullet (mur) management scenarios. The scenarios are defined in Figure 4.2.

4.3.2.3 Thornback ray

The upgrade of the HCR of thornback ray from HCR_{2over3} to HCR_{MSY} had a positive effect on the SSB (Figure 4.12). The use of HCR_{MSY} with default reference point values resulted in a decrease in the probability of the stock SSB falling below safe biological limits by more than 5% compared to the current HCR (HCR_{2over3}), independently of the implementation or not of the landing obligation. Reducing the default fishing mortality reference point or increasing the default biomass reference points in the application of the HCR_{MSY}, the probability decreased by 10%. In this case, the decrease in the probability of the stock SSB was also independent of the implementation of the landing obligation. When the landing obligation was implemented, the probability of the stock SSB falling below the limit SSB was close to zero from 2035 onwards. However, when the landing obligation was not implemented, none of the tested HCRs achieved full protection of the stock, and the probability of the stock SSB falling below safe biological limits was around 15% from 2035 onwards (Figure 4.12).

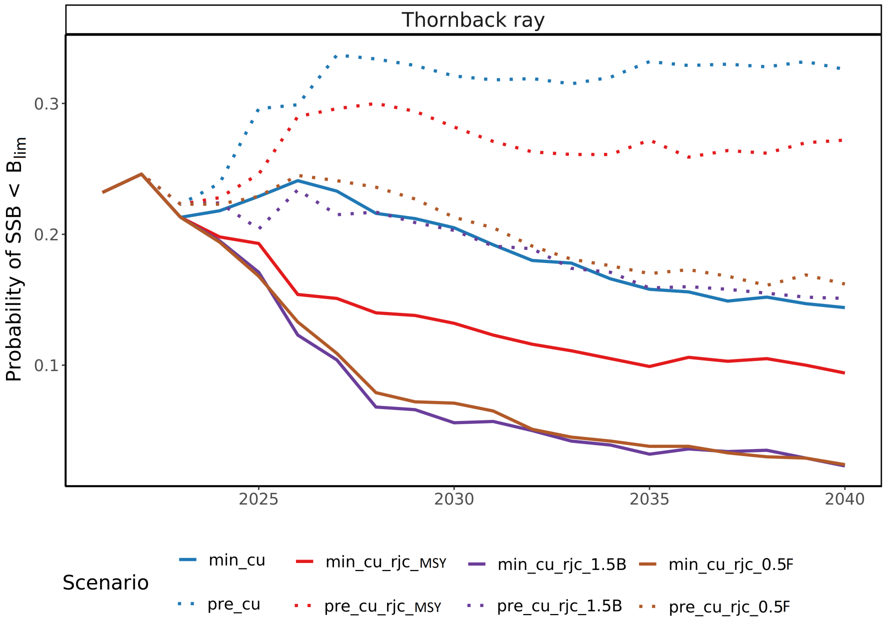


Figure 4.12: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for thornback ray (rjc) management scenarios. The scenarios are defined in Figure 4.2.

When separated TACs for rays and anglerfishes were used, the probability of thornback ray SSB being below B_{lim} was close to zero from 2026 onwards when the landing obligation was implemented (Figure 4.13). However, as showed in the previous section (Figure 4.3), the probabilities with separated and grouped TACs for rays and anglerfishes did not show big differences when the landing obligation was not implemented (Figure 4.12, Figure 4.13).

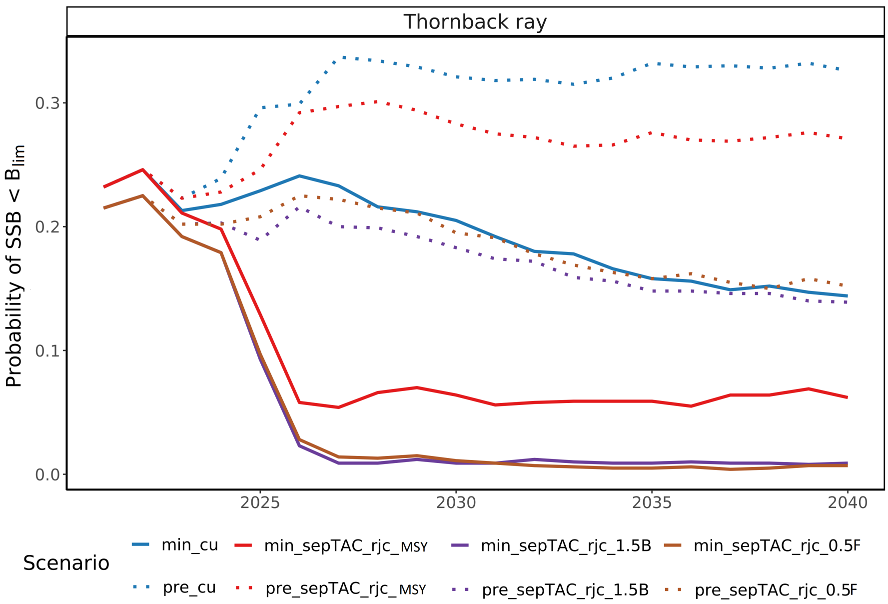


Figure 4.13: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for thornback ray (rjc) management scenarios using separated TAC for each stock. The scenarios are defined in Figure 4.2.

4.3.2.4 Cuckoo ray

The alternative HCRs used to provide management advice of cuckoo ray were not able to guarantee the good state of the stock, and the probability of the stock SSB falling below safe biological limits was higher than 40% in the long-term (Figure 4.14). The probability of cuckoo ray SSB being below B_{lim} increased significantly when the HCR was upgraded from HCR_{2over3} to HCR_{MSY} (Figure 4.14). In fact, the biomass reference points (B_{lim} and MSY $B_{trigger}$) and the fishing mortality reference point (F_{MSY}) were too low and too high respectively (Table 4.4) that the TAC advice given by the HCR_{MSY} was much higher than that given by the current HCR, HCR_{2over3}. The probability of SSB being below B_{lim} decreased when more precautionary biomass and fishing mortality reference points were used in the application of the HCR_{MSY}. However, the probability of SSB being below the limit SSB was still higher than that with the current HCR, HCR_{2over3}. Reducing the default fishing mortality reference point in the application of the HCR_{MSY} resulted in a 10% decrease in the probability of the stock SSB falling below B_{lim} compared to the probability using the default fishing mortality reference points. Increasing the default biomass reference points had a higher positive effect on the SSB and the probability decreased in an additional 20%. This indicates that the default biomass reference point values were more responsible than the default fishing mortality reference point value in giving the high TAC advice for this stock when HCR_{MSY} was used (Figure 4.14). In all the cases, the TAC given by the HCR_{MSY} was that high that there were few differences between the scenarios with and without the implementation of the landing obligation.

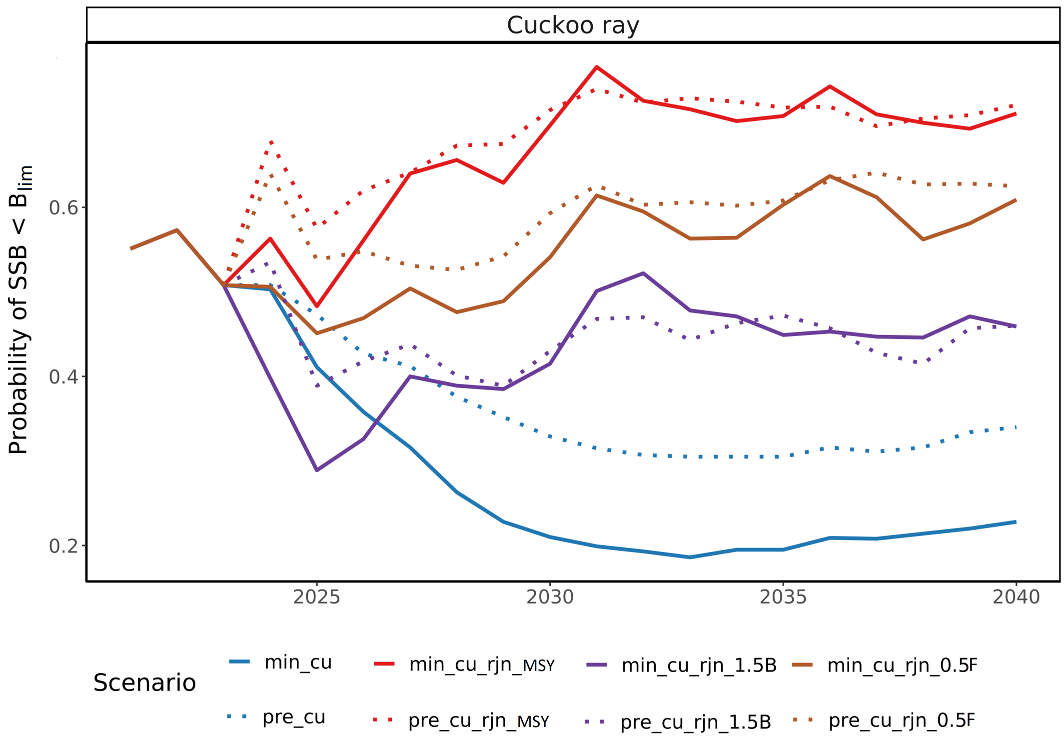


Figure 4.14: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for cuckoo ray (rjn) management scenarios. The scenarios are defined in Figure 4.2.

When separated TACs of rays and anglerfishes were set, the probability of cuckoo ray SSB being below B_{lim} was lower when the landing obligation was implemented than when it was not (Figure 4.15). Overall, the use of HCR_{MSY} resulted in a higher probability of SSB being below the limit SSB compared to the use of HCR_{2over3}. However, under the implementation of the landing obligation, increasing the biomass reference points, the HCR_{MSY} gave a smaller probability of SSB being below B_{lim} than the current HCR (Figure 4.15).

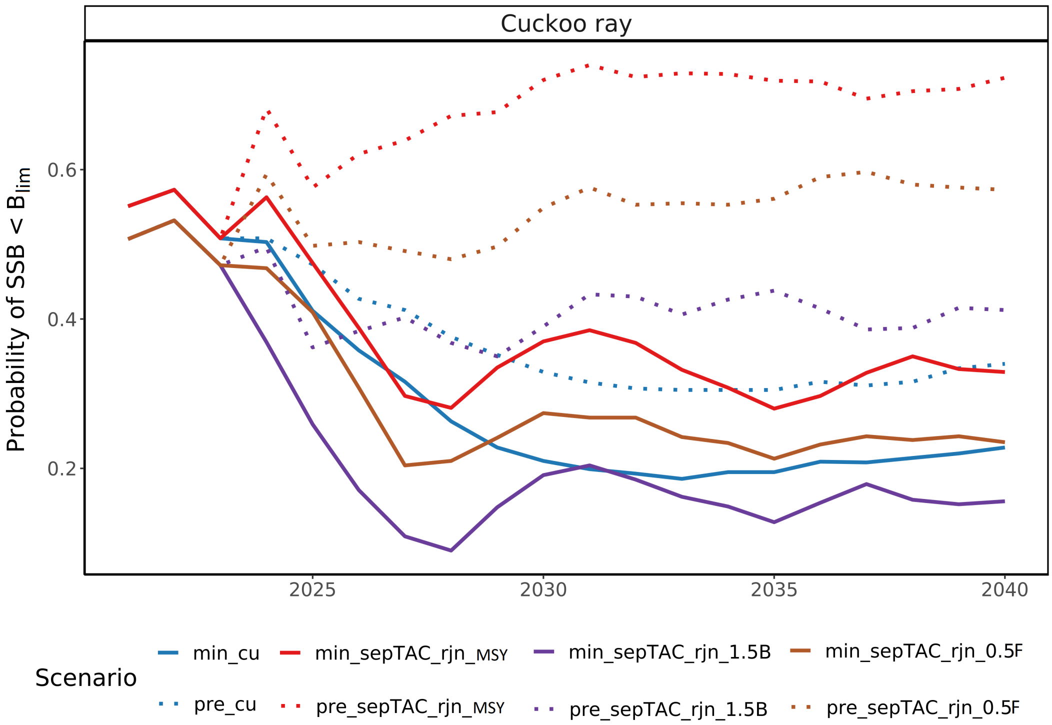


Figure 4.15: Probability of the spawning stock biomass (SSB) falling below the limit SSB (B_{lim}) for cuckoo ray (rjn) management scenarios using separated TAC for each stock. The scenarios are defined in Figure 4.2.

4.3.3 Identification of the factors that have the highest impact on the performance of the management strategy

In this section, we assess the variance of the SSB in the last year of the simulation explained by productivity parameters, starting condition, implementation or not of the landing obligation, HCR used, grouped or separated TACs for rays and anglerfishes and iteration among them.

The SSB was transformed to a logarithmic scale before fitting a linear model to ensure positive response values of SSB. In all the stocks, the SSB in the last year of the simulation did not pass any normality test. The quantile-quantile plot in Figure 4.16 helped to check visually how the lognormal distribution was violated and which data points potentially cause this violation in each data-limited stock. In all the stocks, the data points near the tails do not fall exactly along the straight line (Figure 4.16). In the case of catshark, the assumption that the SSB followed a lognormal distribution was quite appropriate (Figure 4.16). However, for red mullet, thornback ray and cuckoo ray, although the most part appeared to be lognormally distributed, in the right tail of the SSB, the values had a clear departure from the straight line in the quantile-quantile plot (Figure 4.16), indicating that the low values of SSB did not follow a lognormal distribution. In fact, there were several iterations in which the

SSB for those stocks reached zero, and this had an effect on the distribution of SSB. Thus, the results in the right tail of the SSB distribution were not reliable for these stocks.

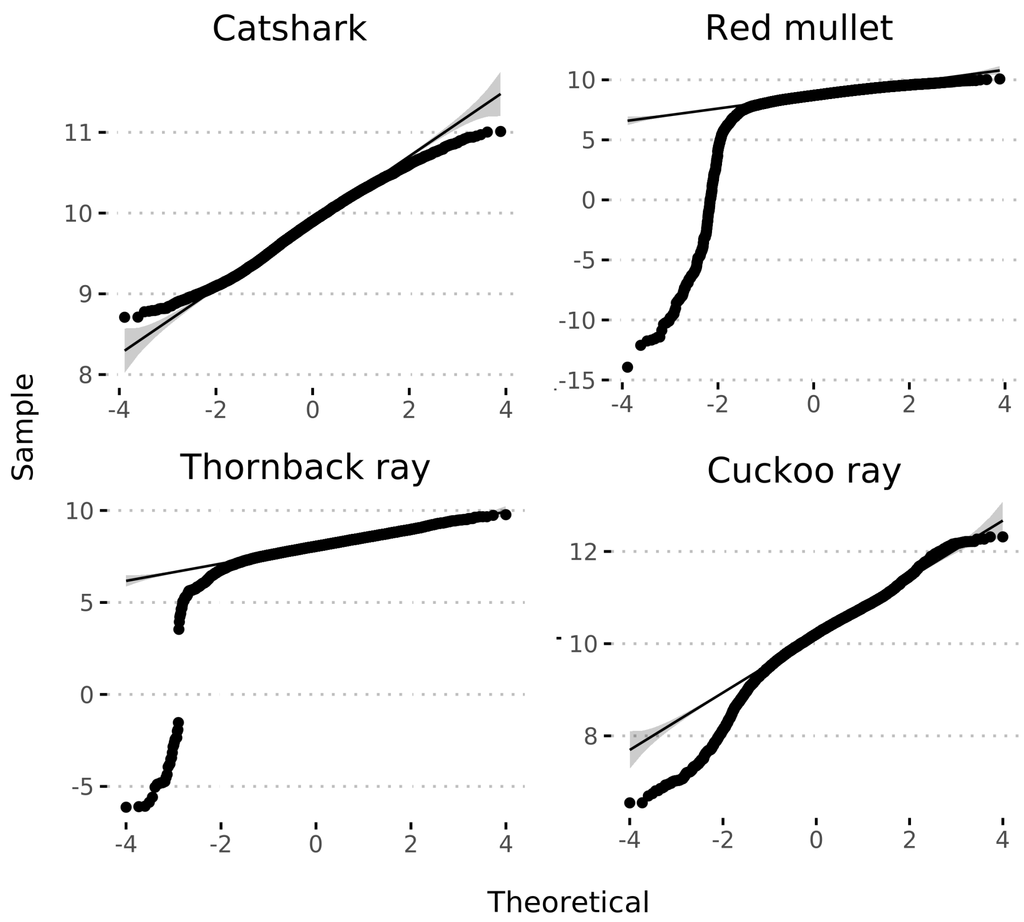


Figure 4.16: Quantile-quantile plot. The lognormal distribution of the spawning stock biomass (SSB) in each of the data-limited stocks. The confidence level of 95% is shaded.

The scatter plots of the productivity parameters and the SSB in 2040, along with their probability distributions, for each management strategy scenario for catshark, red mullet, thornback ray and cuckoo ray are showed in Figure 4.17, Figure 4.18, Figure 4.19 and Figure 4.20, respectively. Overall, the productivity values are widely distributed and fairly scattered throughout the range. Although, for instance in thornback ray, the productivity values concentration was higher at lower values.

The magnitude of the effect of productivity parameters was stock dependent and varied with management strategy scenarios (Figure 4.17, Figure 4.18, Figure 4.19, Figure 4.20). Overall, the higher the productivity parameter value, the higher the SSB in 2040. The only exception was cuckoo ray, for which the performance of the managed system, when its advice was given using the HCR_{MSY} and grouped TACs for rays, was independent of the four productivity parameter values, as shown by the horizontal lines in the fit of the linear model in Figure 4.20. The high catch advice given by the HCR_{MSY} made the SSB of cuckoo ray independent of the productivity parameters.

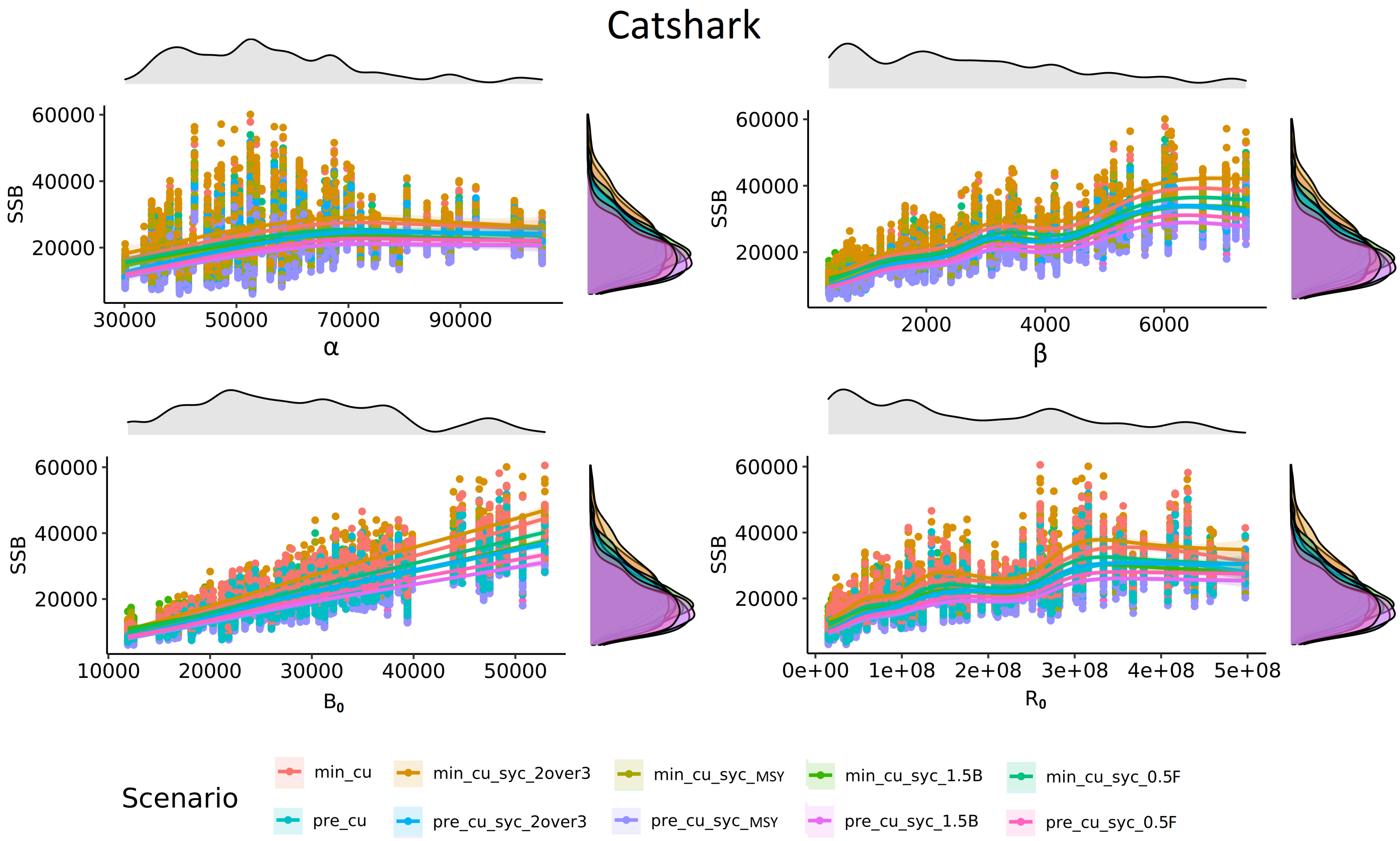


Figure 4.17: Scatter plot of catshark (syc) productivity parameters (α, β, B_0 and R_0) and the spawning stock biomass (SSB, in tonnes) in the last year (2040), along with their marginal probability distributions. The probability distribution of the productivity parameters is on the top of the scatter plot and that of SSB by scenario in the left. Different colors correspond with different management strategy scenarios that are defined in Figure 4.2. The lines correspond with a linear regression to show the trends in the data.

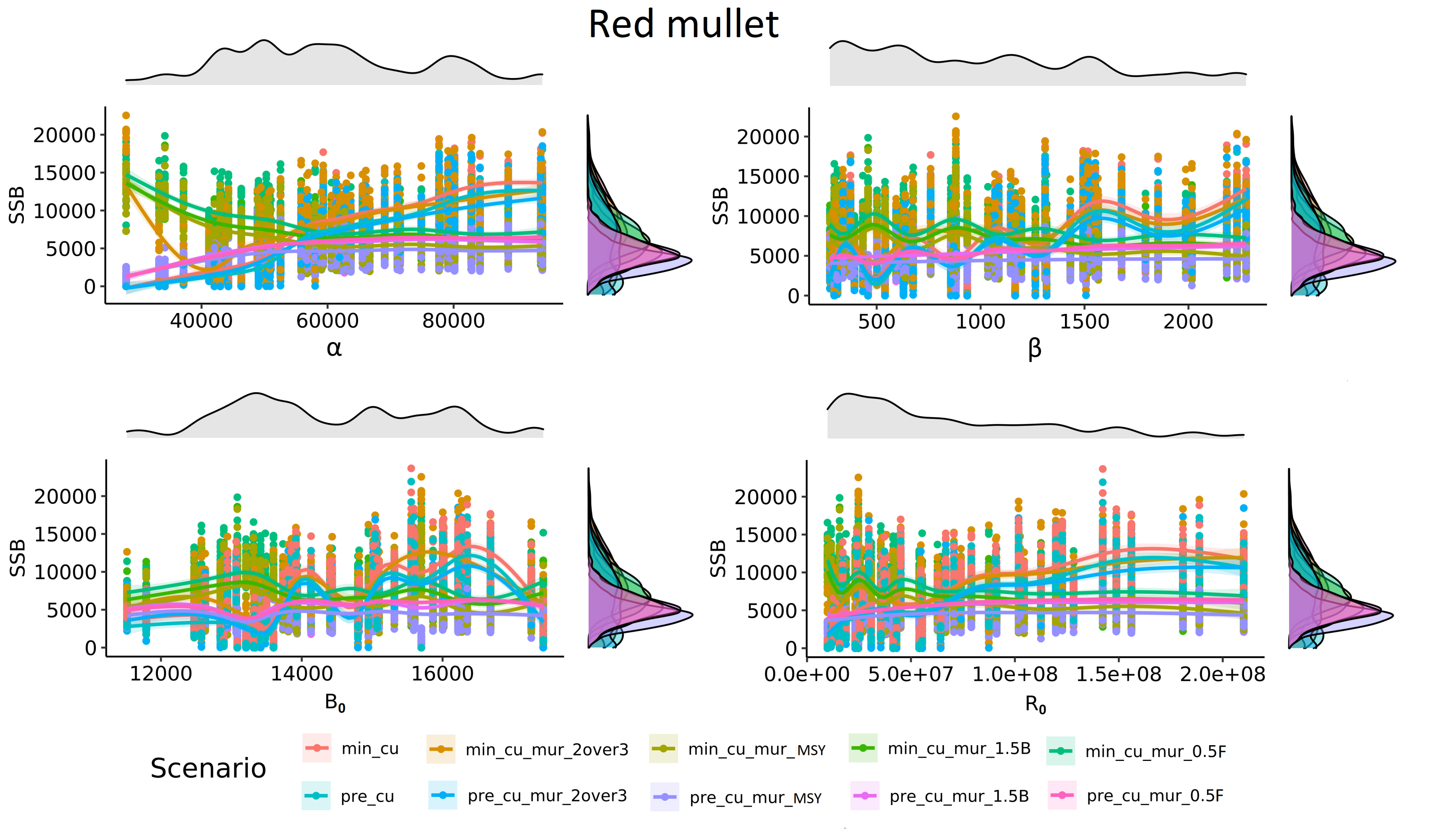


Figure 4.18: Scatter plot of red mullet (mur) productivity parameters (α, β, B_0 and R_0) and the spawning stock biomass (SSB, in tonnes) in the last year (2040), along with their marginal probability distributions. The probability distribution of the productivity parameters is on the top of the scatter plot and that of SSB by scenario in the left. Different colors correspond with different management strategy scenarios that are defined in Figure 4.2. The lines correspond with a linear regression to show the trends in the data.

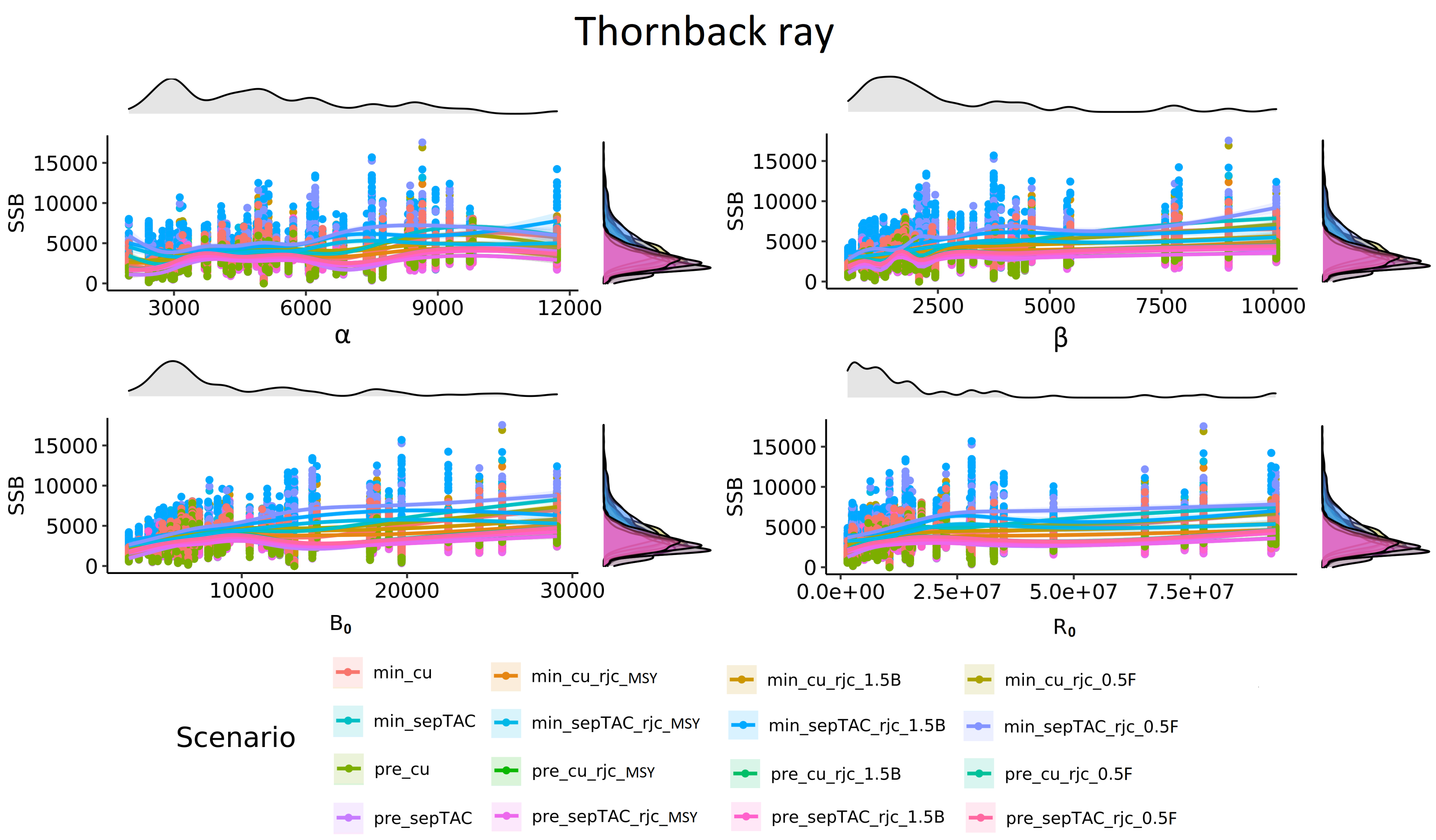


Figure 4.19: Scatter plot of thornback ray (rjc) productivity parameters (α, β, B_0 and R_0) and the spawning stock biomass (SSB, in tonnes) in the last year (2040), along with their marginal probability distributions. The probability distribution of the productivity parameters is on the top of the scatter plot and that of SSB by scenario in the left. Different colors correspond with different management strategy scenarios that are defined in Figure 4.2. The lines correspond with a linear regression to show the trends in the data.

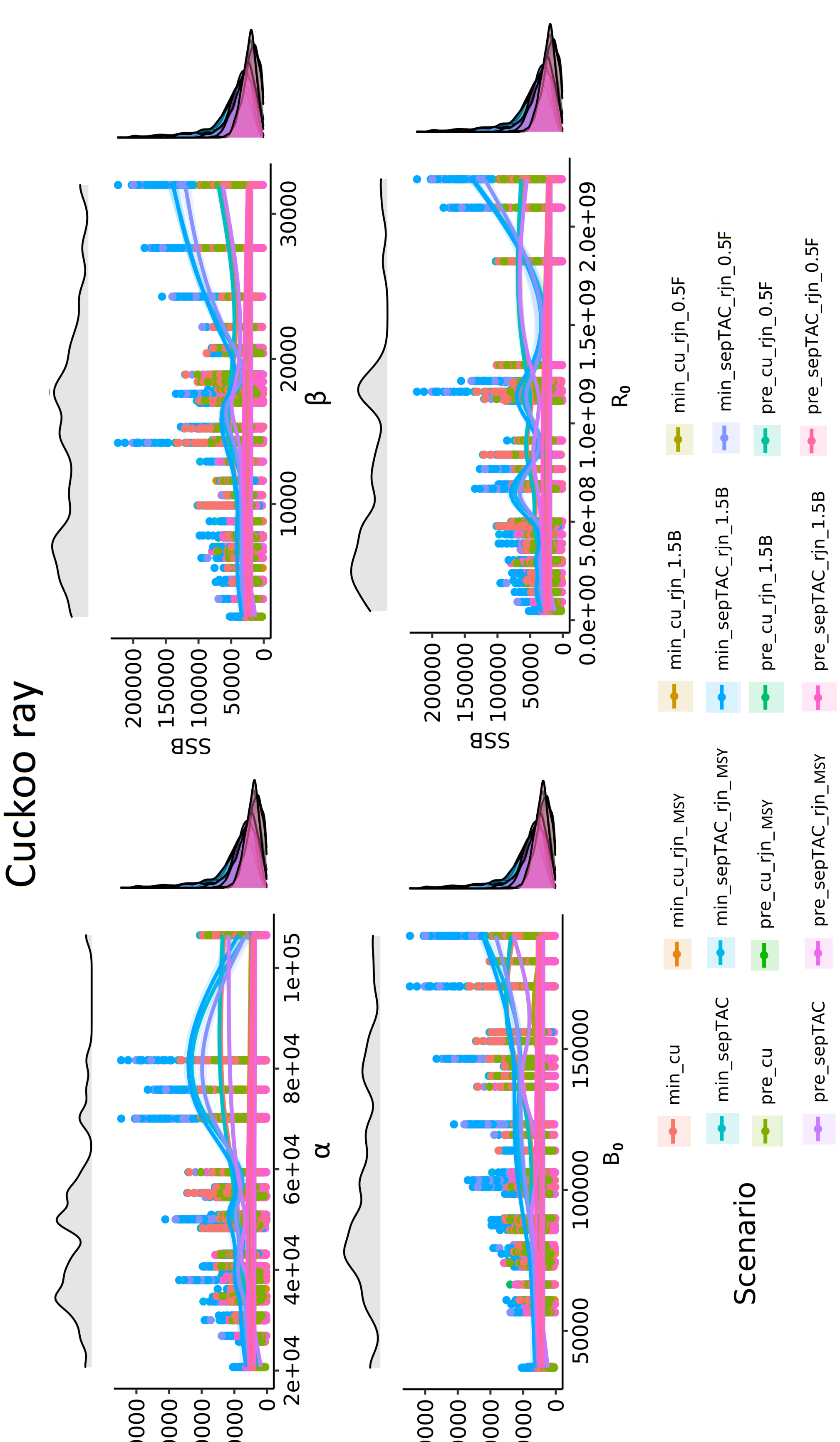


Figure 4.20: Scatter plot of cuckoo ray (rjn) productivity parameters (α, β, B_0 and R_0) and the spawning stock biomass (SSB, in tonnes) in the last year (2040), along with their marginal probability distributions. The probability distribution of the productivity parameters is on the top of the scatter plot and that of SSB by scenario in the left. Different colors correspond with different management strategy scenarios that are defined in Figure 4.2. The lines correspond with a linear regression to show the trends in the data.

The productivity parameters were positively correlated for all the stocks (Figure 4.21). The correlation between both recruitment parameters varied from 0.76 for thornback ray to 0.4 for catshark (Figure 4.21). For all the stocks, the highest correlation was found between β and R_0 productivity parameters (Figure 4.21). That is expected because β controls the shape of the density dependent response of SSB and recruitment, while R_0 is the asymptote recruitment of the curve. The constancy of natural mortality over time further contributes to this correlation between β and R_0. The SSB in the first year of the simulation (2021) was also positively correlated with productivity parameters (Figure 4.21). The parameter with the highest correlation with the SSB in 2021 depended on the stock: for catshark and cuckoo ray was B_0, for red mullet α stock-recruitment parameter, and for thornback ray β stock-recruitment parameter (Figure 4.21).

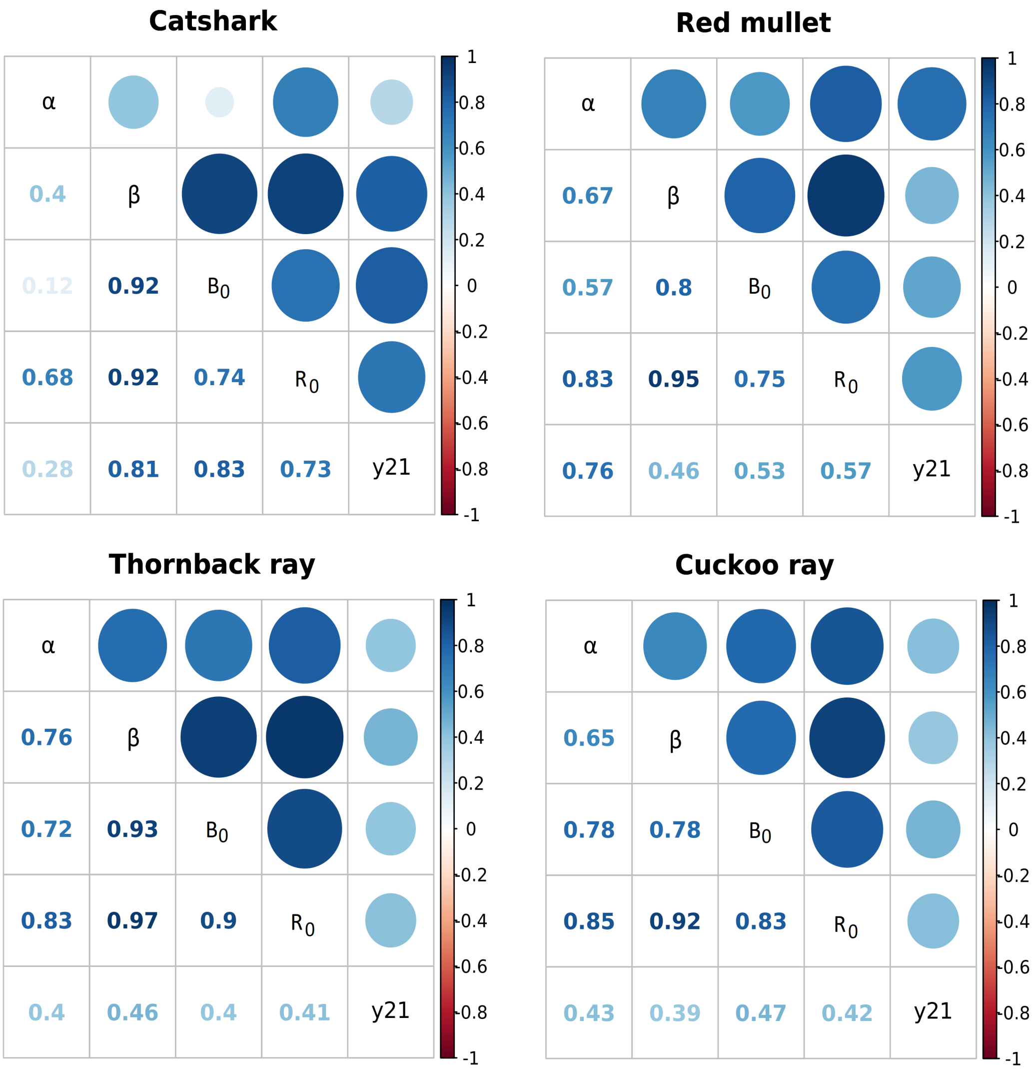


Figure 4.21: Correlation between main productivity parameters (α, β, B_0 and R_0) and the spawning stock biomass (SSB) in the first year of the simulation (y21) for each of the data-limited stocks. Dark blue circles indicate a high positive correlation and dark red values high negative correlation.

The amount of variability in SSB explained by each explanatory variable was stock dependent (Figure 4.22). The explained variance of the SSB by the explanatory variables, was higher for catshark than for the rest of the data-limited stocks. In catshark, the explained variance was 84%, whereas for red mullet, thornback ray and cuckoo ray was 32%, 28% and 20% respectively (Figure 4.22). Among productivity parameters, α and β were those with the highest contribution to the SSB variance (Figure 4.22). For catshark, the contribution of β parameter was 55%, whereas the contribution of α was 12%. For red mullet, the contribution of α was 10% and the contribution of β was 2%. For thornback ray the contribution of α and β were 5% and 2% respectively, and for cuckoo ray was 1% for each. The contribution of B_0 and R_0 was insignificant for red mullet and cuckoo ray, whereas for catshark they explained 10% and 1% of the variance in SSB respectively and for thornback ray explained 3% and 2% respectively. The SSB in 2021 explained 3% for red mullet and 4% for thornback ray. For catshark and cuckoo ray, the SSB in 2021 had not impact on the variance of the last year SSB. The contribution of the HCR was 2% for catshark and 3% for red mullet, thornback ray and cuckoo ray (Figure 4.22). In red mullet, the iteration between the HCR used and the values of α explained 10% of the variance in SSB. The implementation or not of the landing obligation explained 2%, 4%, 5% and 8% of the variance for red mullet, cuckoo ray, catshark and thornback ray respectively. The use of individual or grouped TACs for rays and anglerfishes impacted on the SSB of cuckoo ray where it explains 2% of the variance in SSB. Furthermore, its interaction with the implementation of the landing obligation explained 3% and its interaction with the HCR used explained 1% of cuckoo ray SSB in the last year of the simulation (Figure 4.22).

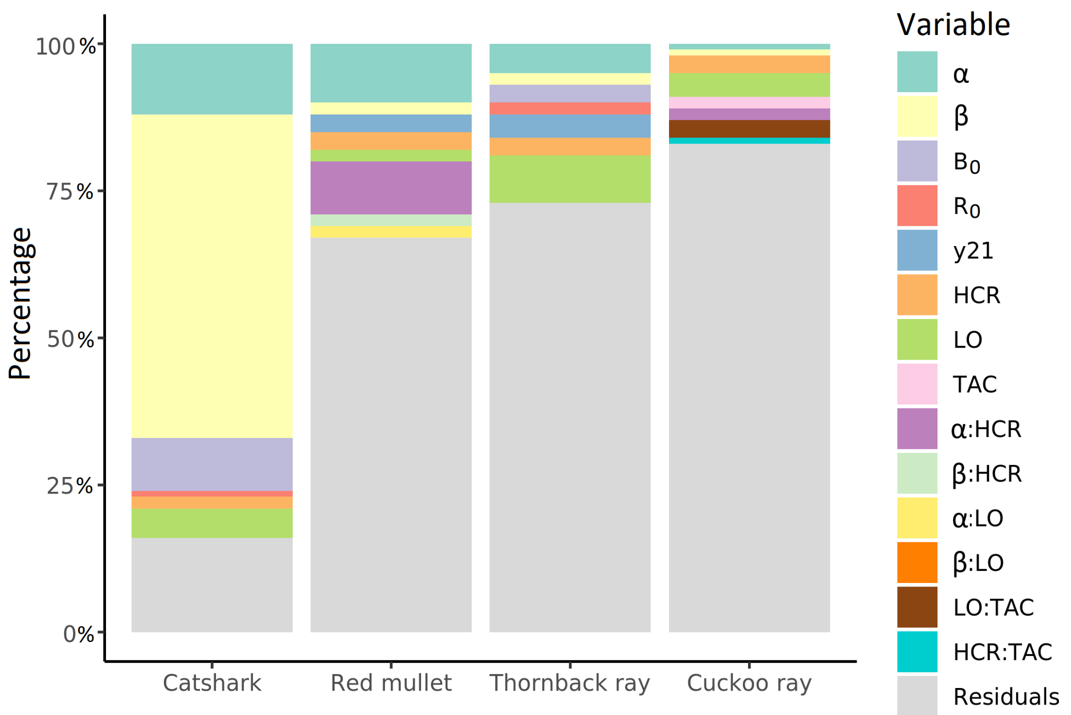


Figure 4.22: Barplot with the variance of the spawning stock biomass (SSB) explained by each variable in the last year of the simulation (2040) for each of the data-limited stocks. The colors correspond with the variables and their interactions.

4.3.4 Evaluation of the proposed management strategy

Based on the results obtained in the previous sections, the stocks that should be included in the management strategy and the best way to provide management advice of each data-limited stock were identified to propose a management strategy that will potentially improve the management of the system. Furthermore, we evaluated how the sustainability of the system improves with respect to the current management strategy.

In the proposed management strategy, the fleets were not constrained by the quota of the pelagic stocks (mackerel, horse mackerel and blue whiting) and the fleets could fish pelagic stocks as much as required, based on the quota share of the other stocks. In fact, the SSB of pelagic stocks was only marginally affected by the management of the demersal mixed-fisheries operating in the Bay of Biscay (Figure 4.3) and they were choke stocks for some of the French fleets (Figure 4.8). In addition, in order to improve the sustainability of red mullet, thornback ray and cuckoo ray, we proposed to include red mullet in the TAC and quota system and provide advice of it using HCR_{MSY} with fishing mortality equal to the half of F_{MSY}, as well as provide management advice of thornback ray using HCR_{MSY} with the biomass reference points equal to the double of default biomass reference points.

Under the landing obligation, the proposed management strategy resulted in an increase in the SSB of the stocks with respect to the current management strategy (Figure 4.23, Figure 4.24). The biggest differences were observed for hake, sole, sea bass, catshark, red mullet, thornback ray and cuckoo ray. However, when the landing obligation was not implemented, the median SSB of all the stocks was similar with the proposed management strategy and with the current management strategy, except for red mullet and thornback ray (Figure 4.23, Figure 4.24). For red mullet, the median SSB was slightly lower with the proposed management strategy than with the current management strategy. However, for thornback ray, the median SSB was higher with the proposed management strategy than with the current management strategy (Figure 4.24). The reason to have lower median SSB for red mullet when the proposed management strategy was used is related to the decrease of uncertainty in the simulated SSB with respect to the current management strategy. In the current management strategy, red mullet was not explicitly managed, so their catches depended on the management measures of the stocks that currently were in the TAC and quota system. This gave a high range of SSB in the simulation period for red mullet (Figure 4.24). However, when in the proposed management strategy red mullet was explicitly managed, its SSB was driven by the management objective of MSY and in the simulation period, the SSB among iterations varied less than when it was not managed (Figure 4.24). Using the proposed management strategy, the uncertainty bound of SSB in the simulation period mainly decreased from high SSB values reducing the median SSB with respect to the current management strategy (Figure 4.24). However, in both stocks, in red mullet and thornback ray, the probability of SSB being below B_{lim}) was lower with the proposed management strategy than with the current management strategy (Figure 4.25).

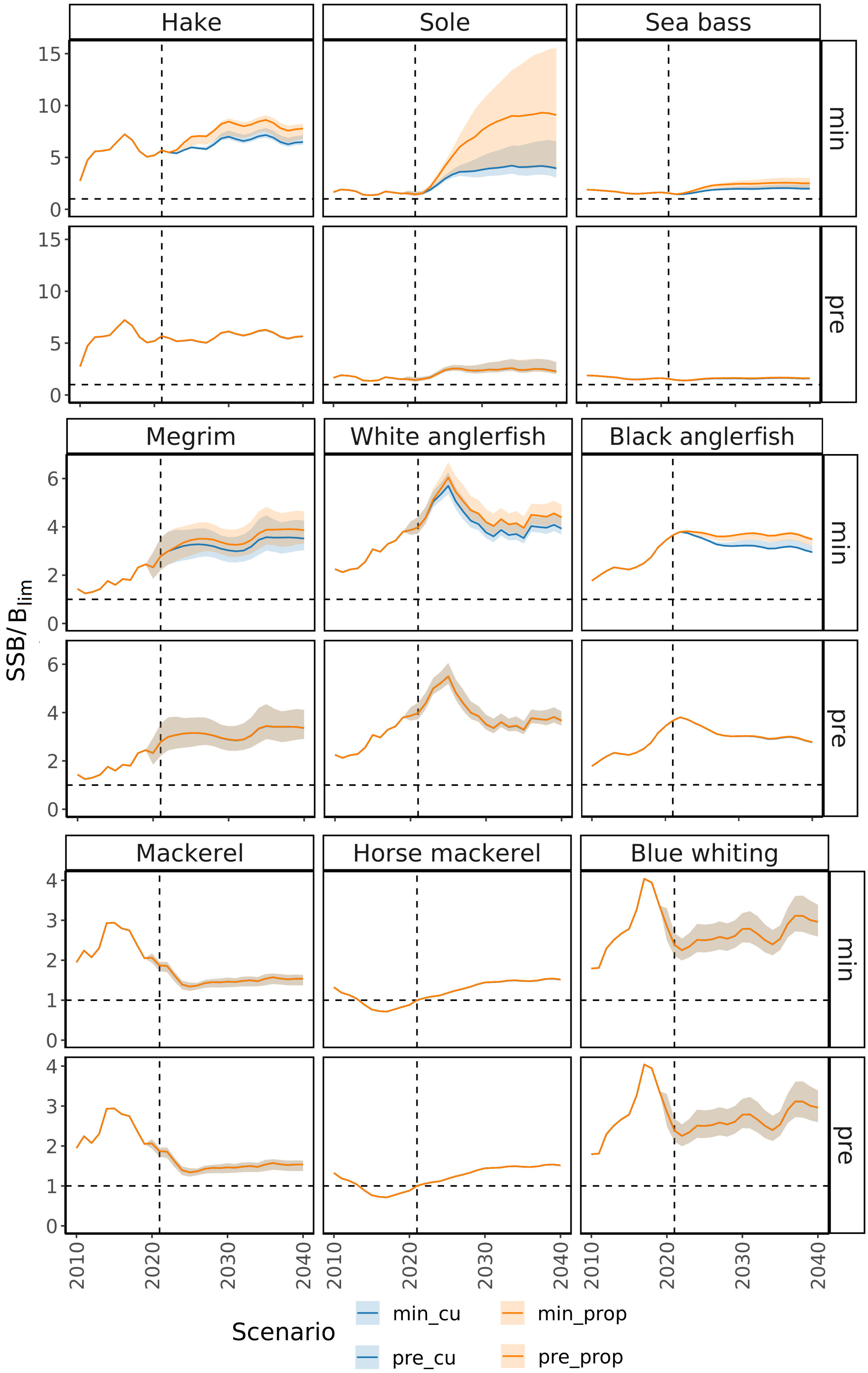


Figure 4.23: Trajectories of the spawning stock biomass (SSB) with respect to the limit SSB (B_{lim}) for data-rich stocks when the current and proposed management strategies were used. Median values (colored lines) and variation (shaded areas; 5% and 95% quantiles) among iterations are shown. The starting year of the simulation (2021) and reference indicator value (SSB/B_{lim} = 1.0) are shown as dashed black lines.

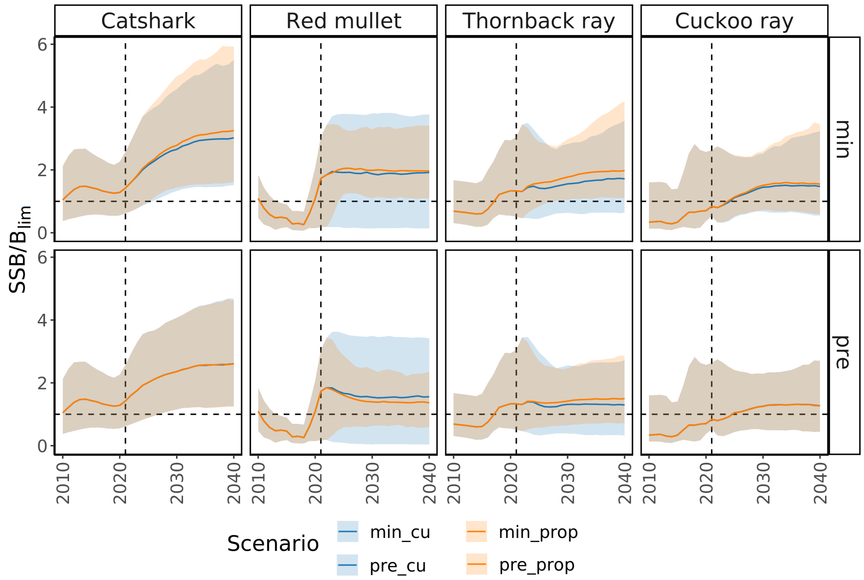


Figure 4.24: Trajectories of the spawning stock biomass (SSB) with respect to the limit SSB (B_{lim}) for data-limited stocks when the current and proposed management strategies were used. Median values (colored lines) and variation (shaded areas; 5% and 95% quantiles) among iterations are shown. The starting year of the simulation (2021) and reference indicator value ($SSB/B_{lim} = 1.0$) are shown as dashed black lines.

Under the landing obligation, the proposed management strategy ensured the good status of all the stocks caught except for cuckoo ray (Figure 4.25). The biggest differences were in the probability of red mullet and thornback ray SSB falling below safe biological limits (Table 4.7). In these cases, with the proposed management strategy the probability was close to zero, whereas with the current management strategy it was around 30% and 15% respectively (Table 4.7). For cuckoo ray the proposed management strategy did not have such a good performance. The probability of SSB being below B_{lim}) decreased only a 3% (Table 4.7).

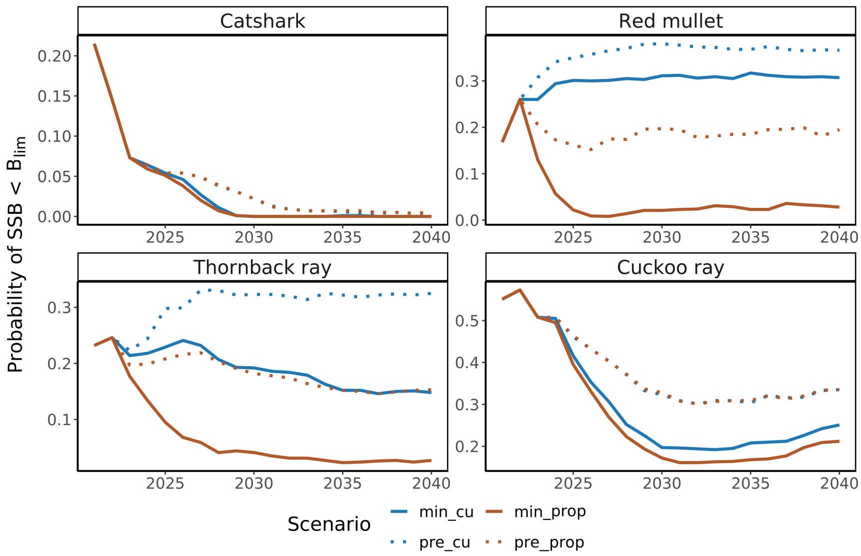


Figure 4.25: Probability of the spawning stock biomass (SSB) being below the limit SSB (B_{lim}) for data-limited stocks when the current and proposed management strategies were used.

When the landing obligation was not implemented, the probability of the stocks SSB falling below safe biological limits also decreased for red mullet and thornback ray when the proposed management strategy was applied (Figure 4.25). For red mullet, the probability of the stock SSB falling below the limit SSB decreased from 37% to 19%, and for thornback ray, from 33% to 15% (Table 4.7). For catshark and cuckoo ray, the probability of the stocks' SSB falling below B_{lim} was similar between the two management strategies (Figure 4.25, Table 4.7). In fact, in the proposed management strategy, the HCRs for both stocks was the same as in the current management strategy, because alternative HCRs tested in this study gave worse or similar performance than the current management strategy.

Table 4.7: Probability of the spawning stock biomass (SSB) being below the limit SSB (B_{lim}) for data-limited stocks in the last year of the simulation (2040) when the current and proposed management strategies were used. The colors indicate the risk levels of the SSB being below B_{lim} (green for low, yellow for medium and red for high probability).

Scenario	Catshark	Red mullet	Thornback ray	Cuckoo ray
min_cu	0.00	0.31	0.14	0.23
min_prop	0.00	0.03	0.02	0.20
pre_cu	0.01	0.37	0.33	0.34
pre_prop	0.01	0.19	0.15	0.34

The differences in catch production between the current and the proposed management strategies were dependent on the fleet (Figure 4.26). Under the implementation of the landing obligation, the catches of most of the fleets were reduced when the proposed management strategy was applied (Figure 4.26). The fishing effort of these fleets was limited by the quota of red mullet, with the probability of red mullet being the choke stock up to 30% (Figure 4.27). Therefore, including red mullet in the TAC and quota system resulted in lower catches in those fleets compared to when red mullet was not included in the TAC and quota system (Figure 4.26). As an exception, the catch of French handliners and French mid water otter trawlers increased with the proposed management strategy (Figure 4.26). In fact, the activity of these fleets was, with the current management strategy, limited by the TACs of pelagic stocks (Figure 4.8), and when the fleets activity was not constrained by the quota of them, the catches in these fleets increased (Figure 4.26).

Overall, without the implementation of the landing obligation, the differences in yield between the current and the proposed management strategies were close to zero or increased with the proposed management strategy by up to 15% (Figure 4.26). The exception were the French handliners, the French mid water otter trawlers with vessel length higher than 24m and the French seiners which lost around 30%, 25% and 20% of the total yield respectively when the proposed management strategy was used (Figure 4.26). The effort of these fleets is mainly driven by the effort needed to achieve the fishing opportunities of pelagic stocks. Thus, when in the proposed management strategy the fishing effort was not updated based on the changes in the quotas of pelagic stocks, the catches decreased with respect to the current management strategy.

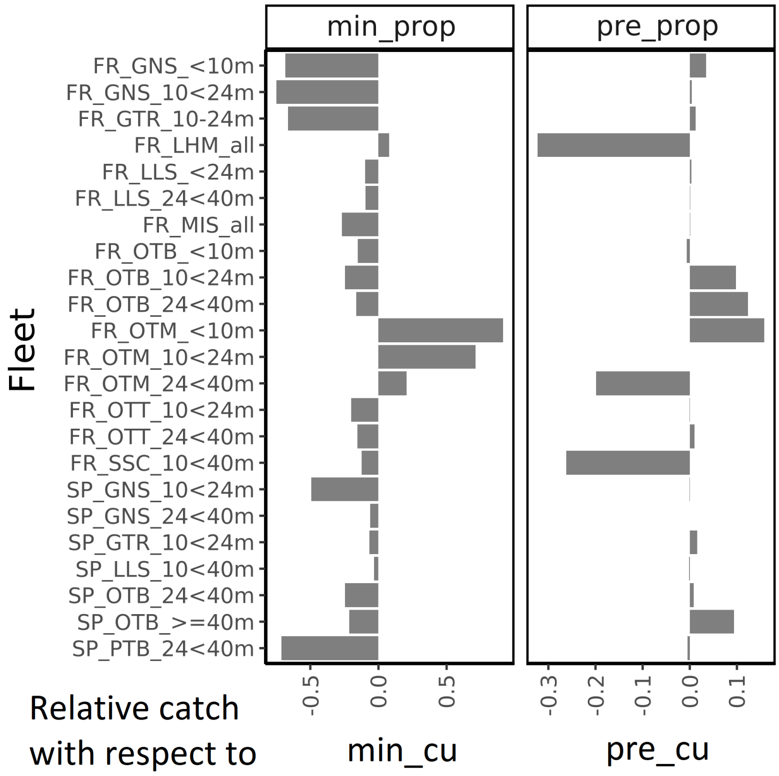


Figure 4.26: Relative difference between the catches of all the stocks over the projection years under proposed management strategy with respect to the currently implemented. Country acronyms: French vessel (FR) and Spanish vessel (SP). Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). The numbers in each fleet name indicates the length of the vessels. The fleets are defined in Table 4.1.

4.4 Discussion

In this chapter, we have implemented a simulation algorithm for the demersal mixed-fisheries operating in the Bay of Biscay. In the simulated reality, the effect of the current and the alternative management strategies on the sustainability of the most important stocks, and on the performance of the fishing activity, in terms of yield, was evaluated. The results indicated that the mixed-fisheries multiannual management plan currently in place (EU, 2019b) is not able to guarantee the sustainability of red mullet, thornback ray and cuckoo ray in the short- and long-term. The sustainability of the system improved when red mullet was included in the TAC and quota system and when the HCR of thornback ray was upgraded from a HCR based on an abundance index to the MSY advice rule. In addition, the analysis of the contribution of productivity parameters, starting condition and management strategy attributes to the variability in the last year SSB showed that overall, the stock-recruitment relationship parameters, were the ones that have the highest impact on the performance of the management strategies. Additionally, the implementation of the landing obligation also had a high impact on the performance of the management strategies, and the type of HCR contributed highly to explain the variability of the last year SSB.

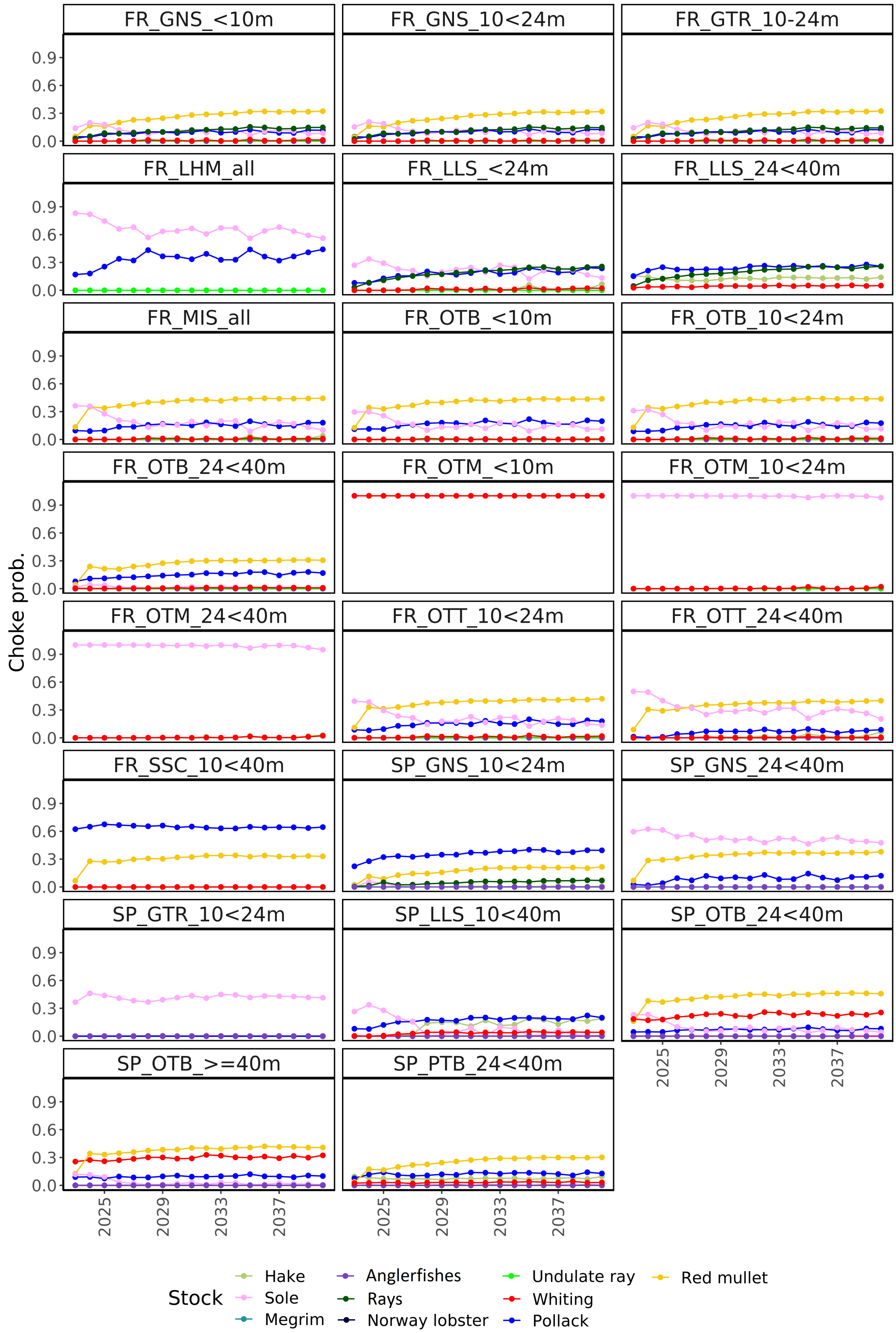


Figure 4.27: For each fleet, the probability of the effort being limited by each stock in each year of the simulation considering the proposed management strategy and under the landing obligation. Country acronyms: French vessel (FR) and Spanish vessel (SP). Gear acronyms: gillnetters (GNS), trammel nets (GTR), handliners (LHM), longliners (LLS), miscellaneous gear (MIS), bottom otter trawlers (OTB), mid water otter trawlers (OTM), twin-rigged otter trawlers (OTT), seiners (SSC) and bottom pair trawlers (PTB). The numbers in fleet's name indicates the length of the vessels. The fleets are defined in Table 4.1.

The demersal mixed-fisheries that operates in the Bay of Biscay catch more than 150 species (Altuna-Etxabe et al., 2020; Briton et al., 2020). However, the management of the fishery is focused on approximately one-fourth of these species (Briton et al., 2020) and most of them are caught without any quota restriction (EU, 2022). The STECF in 2015 (STECF, 2015) evaluated the potential impact of the Western Waters and adjacent waters multiannual management plan on the demersal mixed-fisheries operating in the Bay of Biscay. They used two mixed-fisheries simulation model implementations, one focused on the French fleets and other on the Spanish fleets. In the former, 17 species where included and only three of them were simulated using an age-structured population dynamics model, whereas in the later, 12 species were included and only two of them were simulated using an age-structured population dynamics model. Comparing with the evaluation of the STECF (2015), in this study, the French and Spanish fleets dynamics were conditioned in the same operating model and the number of species included was extended to 28 species. Additionally, the number of species included dynamically in the model increased to 13, where four of them are data-limited (catshark, red mullet, thornback ray and cuckoo ray). The population dynamics of most of the species that are included in the TAC and quota system and in consequence could limit the fishing activity of the demersal mixed-fisheries in the Bay of Biscay, were included in this simulation study.

The current management plan, focused on the management of eight data-rich stocks, ensures the sustainability of all data-rich stocks and the sustainability of one of the four data-limited stocks included in the simulation, namely, catshark. On the contrary, it does not ensure the sustainability of red mullet, thornback ray and cuckoo ray. Red mullet is not included in the TAC and quota system, however, we found that to ensure its sustainability it is necessary to introduce specific management measures for the stock. In the case of thornback ray and cuckoo ray, both are in the TAC and quota system, however, the HCR based on an abundance index used to provide management advice for them was not able to guarantee the good status of rays. Additionally, the sustainability of other stocks not assessed in this simulation study, whether excluded from the simulation or included with a constant CPUE approach, could also be compromised under the current management plan.

The sustainability of the system, mainly improved when red mullet was included in the TAC and quota system using the MSY advice rule with fishing mortality lower than the MSY value and when the HCR of thornback ray was upgraded from the HCR based on an abundance index to the MSY advice rule with the biomass reference points higher than the default biomass reference points. Currently, the data availability of those stocks is not sufficient to upgrade them to the ICES category 1, however, giving their fishing advice based on MSY advice rule was more precautionary than using a HCR based on an abundance index. In 2022 ICES defined a new framework with new HCRs to provide management advice for stocks in ICES category 3 (ICES, 2022a). Previous findings indicated that the HCR tested in this chapter and used by ICES for stocks in ICES category 3 was not always precautionary (Fischer et al., 2020, 2022a). The new HCRs defined are consistent with precautionary approach and two of them are also consistent with MSY approach (ICES, 2022a). In addition to considering the biomass index trend and the catch trend, the new HCRs consistent with the MSY approach utilize length and growth data to guide catch advice (ICES, 2022a). Fischer et al. (2022a) concluded that simple modifications of the previous HCRs, such as including a multiplier to guide fishing pressure, can be sufficient to obtain desired management objectives. The new HCRs were not tested in this study, however, it must be interesting in any future work to explore their performance in the case study to provide advice of data-limited stocks. Overall, as expected, the HCR based on an abundance index performed worse than the MSY advice rule. Thus, as showed recently by Fischer et al. (2022b) the HCR based on an abundance index should be phased out because it is not fit for purpose of achieving long-term management objectives. As an exception, the MSY advice rule performed worse than the HCR based on an abundance index for cuckoo ray and led to non-precautionary management of the stock. The same was found previously for European plaice, Atlantic cod and Atlantic herring in Fischer et al. (2022b) and for North Sea whiting and herring in ICES (2019c). Apparently, this was caused more by how the MSY advice rule was applied than by the formulation of the MSY advice rule itself (Fischer et al., 2022b). In this case, the biomass and fishing mortality reference points used to apply the MSY advice rule resulted to be too low and too high respectively, and the TAC advice given by the MSY advice rule led to higher fishing opportunities than the HCR based on an abundance index in cuckoo ray.

The sustainability of data-limited stocks, when the proposed management strategy was implemented, was highly dependent on the implementation of the landing obligation. When the landing obligation was implemented, the proposed management strategy guaranteed the sustainability of red mullet and thornback ray and slightly reduced the probability of cuckoo ray SSB being below B_{lim} in the long-term. The implementation of the landing obligation also resulted in an increase of the SSB of some data-rich stocks in relation to the current management plan. However, in terms of fishing yield, the inclusion of red mullet in the TAC and quota system limited the fishing effort of most of the fleets which supposed a significant loss in yield. When the landing obligation was not implemented, the probability of red mullet and thornback ray SSB being below B_{lim} in the long-term decreased by half with respect to the current management plan, whereas the probability for cuckoo ray and the rest of the stocks was equal to that in the current management plan. The total yield of the fishery remained unchanged when comparing the proposed management strategy to the current management plan.

None of the management strategies tested in this study could guarantee the full protection of cuckoo ray in the long-term. This limitation was due to the low biomass level of cuckoo ray observed in the initial year of the simulation. Demersal stocks, including cuckoo ray, are characterized by low productivity, long lifespan, and late maturity, resulting in a reduced capacity to recover from low biomass levels. Specifically, for cuckoo ray, there was a probability of over 50% that the SSB would fall below B_{lim} in the first year of the simulation. Consequently, the recovery of this stock needs a more tailored and specific management strategy beyond the scope of those analysed in this chapter. Conducting a spatio-temporal analysis of the cuckoo ray population and fisheries distribution would help identify areas of overlap and establish potential spatio-temporal management measures, such as targeted fishery closures in specific locations and time periods.

The correct implementation or not of landing obligation was simulated using two different fleet dynamics scenarios because there are doubts about the compliance of the landing obligation (Prellezo and Villasante, 2023). The results depended greatly on this assumption. In mixed-fisheries, enforcing the landing obligation would safeguard the stocks' sustainability but would also lead to a strong reduction in fishing effort due to the choke effect, and subsequently, most of the stocks quota would remain underutilized, resulting in significant social and economic negative impact (Rindorf et al., 2021). In the scenario where the landing obligation was not implemented, the yearly effort depended on both, the previous years' fishing effort and the efforts corresponding to each of the quotas of the stocks caught by the fleet. This might be closer to the real fleet dynamics, however, the profits obtained from the activity and the tradition are factors that have been proven to drive fleet dynamics (Marchal et al., 2013; Marchal and Vermard, 2022; Simons et al., 2014). In this case, tradition has somewhat considered comparing the effort to the previous effort level, but more tailored approach could describe better the fleet dynamics. In further analysis, it would be convenient to consider them as factors that drives the fleet dynamics in order to obtains a better description of fishing activity and hence increase the accuracy of the results obtained.

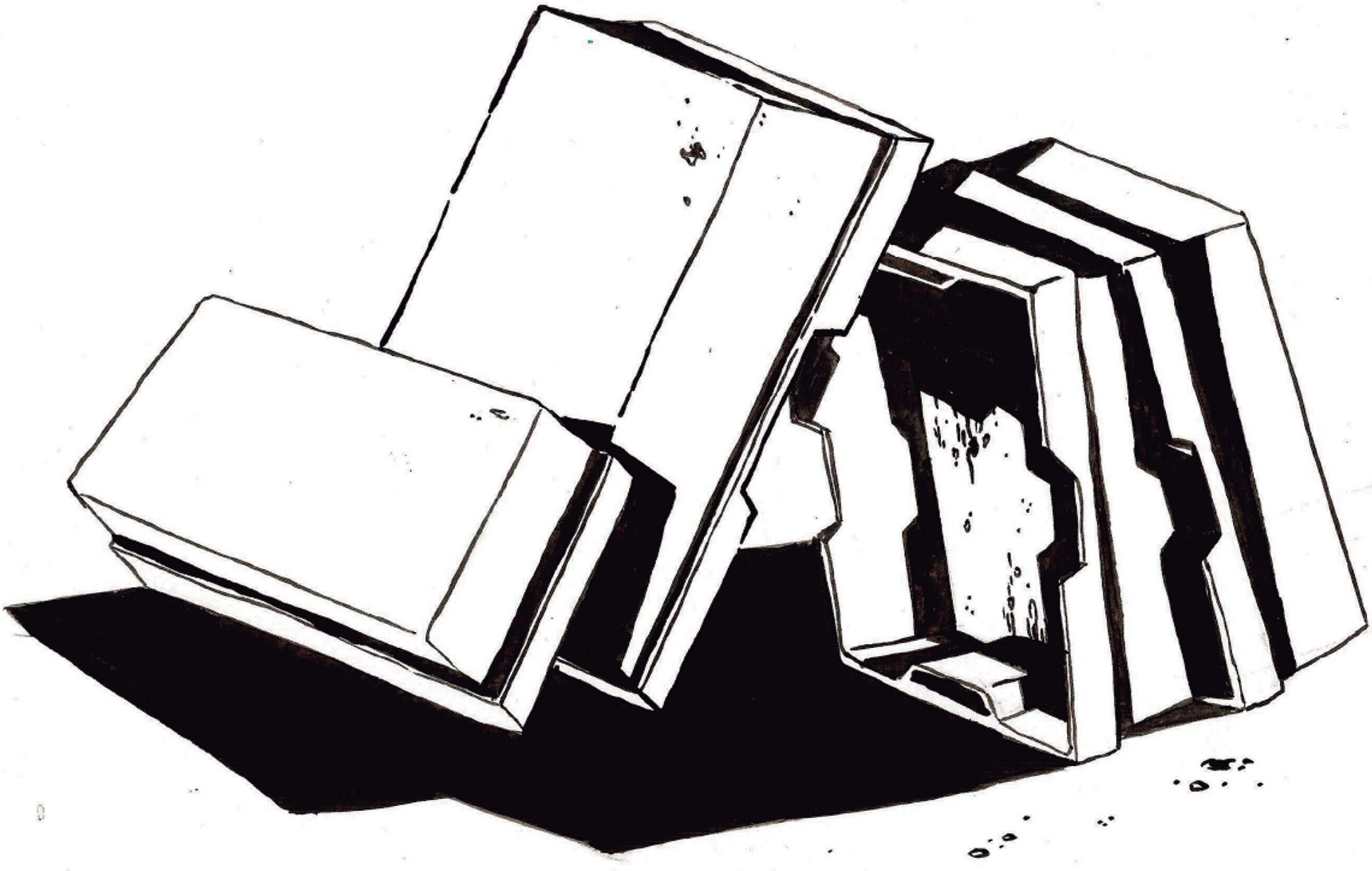
Currently, fishing opportunities for rays and anglerfishes are provided through grouped TAC at genus level, as it is challenging to differentiate between stocks within the same genus. We found that while the grouped TACs promotes the stability of the fishing activity, it could compromise the sustainability of the stocks. When individual TACs for rays and anglerfishes were used, the TAC of thornback ray limited the fishing activity of most of the fleets and produced a misuse of other stocks fishing opportunities that was alleviated when grouped TAC was used. The use of grouped TACs resulted in fishing levels higher than the recommended at single stock level, and it had a negative impact on the rays and anglerfishes SSBs.

It is well known that stock-by-stock management objectives are not reached for all the stocks simultaneously in mixed-fisheries (Ulrich et al., 2011). In fact, in mixed-fisheries, several stocks with different productivity and susceptibility are caught together without the possibility of discriminating among them (Ulrich et al., 2016). Fishing mortality ranges where introduced in mixed-fisheries multiannual management plans to provide flexibility to the TAC and quota system. The upper and lower bounds of the F_{MSY} range are defined to deliver no more than a 5% reduction in long-term yield, compared with the MSY obtained by fishing at F_{MSY} in the long-term (ICES, 2015b). The objective of establishing a fishing mortality range is to alleviate the choke effect experienced by certain stocks under the landing obligation (EU, 2014; ICES, 2015b). In our case study, the impact of fishing mortality ranges is limited as the multiannual management plan only considers eight stocks caught by demersal mixed-fisheries operating in the Bay of Biscay. However, there are additional stocks, such as data-limited stocks (i.e., pollack, thornback ray, cuckoo ray and undulate ray) and pelagic stocks (i.e., horse mackerel, mackerel, and blue whiting), which are subject to the TAC and quota system and in consequence to the landing obligation (EU, 2022). Particularly, pollack, thornback ray and horse mackerel have a high likelihood of constraining fishing efforts in the demersal mixed-fisheries operating in the Bay of Biscay. However, unfortunately, these stocks are not currently included in the multiannual management plan (EU, 2019b), which means that their catch advice does not include any flexibility. Flexibility should be introduced in the TAC and quota of all the stocks to enhance the overall flexibility of the system.

The fishing mortality ranges were not tested in this simulation study. In STECF (2015) the upper and lower bounds of fishing mortality around F_{MSY} were evaluated with respect to the use of F_{MSY}. However, these scenarios were not realistic: in the same way that F_{MSY} cannot be simultaneously reached for all stocks, neither can the upper and lower bounds of F_{MSY}. Thus, a mechanism to operationalize the ranges in practice is required. Ulrich et al. (2016), García et al. (2019) and Briton et al. (2020) explored operational options to reconcile single-stock management objectives in the mixed-fisheries context. Ulrich et al. (2016) proposes an approach that provides an optimal set of fishing mortalities within MSY fishing mortality ranges, minimizing the differences between the catch advice obtained in two contrasting fleet dynamics scenarios; a scenario where the fleets stop fishing when the first quota is consumed, and an scenario where they do not stop fishing until the last quota is consumed. The major advantage of this approach is that it provides optimal catch shares at fleet and/or country level. The approach proposed in Ulrich et al. (2016) was only applied to mixed-fisheries data and its performance was not evaluated in a simulated reality. Thus, it would be necessary to analyse its performance in a simulated reality to evaluate if in practice it helps to reduce choking effect, resulting in a better use of fishing opportunities. In the approach proposed by García et al. (2019) a linear relationship between fishing mortality and fishing effort is assumed, and they use a multiplier to the fishing mortality to maximize the fishing opportunities while keeping the advice fishing mortality below the upper bound of the MSY fishing mortality ranges for all the stocks. The low data requirement of the approach proposed by García et al. (2019) is one of the main advantages compared to the one proposed by Ulrich et al. (2016). In fact, the approach proposed by García et al. (2019) can be applied using only the output data from stock assessment models, making it easily applicable on a regular basis to generate catch advice. The performance of the approach was tested in a simulated reality. Its main limitation is that in multi-fleet cases, the HCR can only work effectively if the catch profile of the fleets is similar to the overall catch profile. This could also happen in the approach proposed by Ulrich et al. (2016) if the resulting catch advices are share among fleets according to historical shares and not to their catch profiles. In the approach proposed by Briton et al. (2020) the fishing mortality for each stock is sampled from the range of 0.1 to 0.8. These fishing mortalities were subsequently utilized in the simulation model to explore the relationship between different stocks fishing mortalities, considering technical interactions between them. Among the numerous fishing mortality combinations, those that guarantee the utilization of at least 95% of the quota for each stock were selected. Afterward, the performance of each fishing mortality combination in meeting the pre-specified economic, social, and ecological objectives is evaluated in a simulated reality. The main advantage of this approach is that the trade-offs between different objectives are transparent because all the possible fishing mortality combinations that consumed 95% of all the stocks quotas are tested. The main limitation of this approach is that it is designed in two dimensions, i.e., it is based on how the fishing mortalities of two stocks are reconciled based on technical interactions, and the methodology lacks operability for fisheries with more than three interacting stocks, which is the case for most mixed-fisheries. Indeed, as more stocks are included in the analysis, the number of possible combinations of fishing mortalities increases. This expansion of potential scenarios and outcomes can make the interpretation of the results more complex and challenging. With a larger number of stocks, the interactions and trade-offs between different fishing mortalities become more intricate, requiring a more comprehensive and nuanced understanding of the system. These three approaches offer objective and mechanistic ways to generate yearly catch advice within the MSY fishing mortality ranges, avoiding biases towards either end of the bounds. They demonstrate the importance of considering technical interactions between stocks, advice flexibility, adaptive management, and stakeholder engagement in the development of management approaches for mixed-fisheries to achieve sustainable yields while accounting for ecological objectives. Applying these approaches within our simulation framework would be of great interest. However, before applying these approaches in our case study, it is important to consider certain factors that can improve their reliability and performance in the context of a multi-fleet and multi-stock fishery. Firstly, the approach proposed by Ulrich et al. (2016) should be tested within an MSE framework to obtain a realistic understanding of its applicability in a real system. This testing would provide valuable insights into how well the approach performs and whether any adjustments are necessary. Secondly, the approach proposed by García et al. (2019) would benefit from being applied at the fleet level. This is because each fleet within our fishery exhibits a different catch profile that deviates from the overall catch profile of the fishery. By considering the individual catch profiles of each fleet and applying the approach at the fleet level, we can achieve a more accurate representation of the fishery dynamics. To implement this, catch and effort data at both fleet and metier levels, as used in the approach proposed by Ulrich et al. (2016), would be required. The multiplier used in García et al. (2019) would be applied to the reference effort level. However, it should be noted that applying the model at the fleet level instead of the stock level, as currently done, may result in the loss of operationalization of the results. Assumptions and transformations would need to be made to convert fleet level results into catch advice for individual stocks. Lastly, the approach proposed by Briton et al. (2020) needs to be generalized to be applicable to real mixed-fishery situations involving several stocks. To address the issue of multidimensionality, it is necessary to analyse the technical relationship between fishing mortalities of all the stocks. This could be achieved by optimizing the exploration of ecoviable strategies, similar to the approach developed by Gourguet et al. (2013), rather than relying on the exhaustive screening of all possible combinations as done in Briton et al. (2020). For the operationalization of the fishing mortality ranges it is crucial to allocate fishing quotas across the fleets according to their catch profiles in order to prevent choking effects and ensure the good utilization of quotas (García et al., 2019). As seen, there are several approaches available to generate yearly catch advice within the MSY fishing mortality ranges. However, each approach relies on various assumptions, which can lead to different outcomes in terms of determining the catch advice.

One of the challenges when implementing a simulation model is the conditioning of the system dynamics and the identification of the main sources of the uncertainty (Punt et al., 2016). The population dynamics of data-rich stocks in our study were simulated using the most recent assessments available. For some data-limited stocks, however, the population dynamics were simulated using the SRA model (Hordyk et al., 2021). The conditioning of uncertainty is one of the main limitations in the stock assessment (Ducharme-Barth and Vincent, 2022; Rosenberg and Restrepo, 2011). The SRA model is a valuable tool for integrating key uncertainties of data-limited stocks dynamics within the assessment model framework. The SRA model permit to incorporate uncertainty in all the life-history parameters and catch and abundance index data. In this study, the population dynamics of data-limited stocks were defined including uncertainty in all the life-history parameters and catch and abundance index data. The inclusion of uncertainty in data-limited stocks resulted in wide confidence intervals for population dynamics and exploitation level estimates. However, it is important to note that the uncertainty included in the analysis was not based on real data, which raises the possibility that the output uncertainty might be misrepresented. In fact, the "true" uncertainty is unknown and could potentially differ from what was assumed in this study. Attempts were made to incorporate higher uncertainties, but it yielded implausible population dynamics estimates, such as extremely high R_0 values, indicating limitations in including greater uncertainty in the life-history parameters. This could be related to the fact that correlation between the values in life-history parameters was not considered when introducing uncertainty. For both data-rich and data-limited stocks, the simulation model accounted for the process and parameter uncertainty associated with their respective population dynamics. For that, the population dynamics estimates of data-rich stocks were included in the simulation model introducing uncertainty in the assessment model's estimates of natural mortality, weight-at-age, maturity-at-age, and initial numbers-at-age. This uncertainty was taken into account only if it was included in the calculation and evaluation of reference points when performing stochastic simulations. Additionally, parametric uncertainty and random variability around point estimates were considered when conducting the simulation. For data-limited stocks, a set of OMs derived from the SRA model were used to condition their population dynamics in the simulation model. The observation uncertainty was only included in the historical period of data-limited stock population dynamics. The implementation error emerged naturally from simulation of mixed-fisheries dynamics. In both fleet dynamics scenarios, the catches of all the stocks did not fully align with the advised catch. Under the landing obligation scenario, the advised catch of the most restrictive stock was reached before the fishing activity stopped resulting in lower catches than the advised catch for the remaining stocks. In the scenario based on previous dynamics, catches exhibited variations across different stocks, with some caught below their advised catch levels, some caught at their advised catch levels, and others exceeding their advised catch levels. We include two-year time lag between the observed population and the implementation of the advice, reflecting the current practice in fisheries management for these stocks. However, it is important to note that we did not directly include uncertainties related to observation (e.g., inadequate data collection systems), assessment model (e.g., equations or models used to describe the population dynamics), and institutional factors (resulting from the interactions among stakeholders in the management process). These uncertainties, along with process, parameter and implementation uncertainties, are also recognized in Francis and Shotton (2011) as important sources of risk in fisheries settings. The accurate representation of uncertainty is crucial for the effectiveness of MSE in achieving fisheries management objectives (Punt et al., 2016). Punt et al. (2016) identified that minimally, a MSE simulation model should include process, parameter and observation uncertainty. Future work should focus on a good characterization of the input uncertainty.

The SRA model admits including uncertainty in all the parameters and data used, and requires little data, which is especially relevant for data-limited stocks. Unfortunately, the data available for most of the data-limited stocks included in the simulation framework was limited to short-catch time-series which precludes the implementation of the SRA model. Their population dynamics could not be modelled reliably by the SRA model and their conditioning in the simulation algorithm had to be based on the constant CPUE approach. Among these data-limited stocks, pollack, withing and undulate ray are in the TAC and quota system, and the first two stocks had high probability to limit the fishing effort of most of the demersal mixed-fisheries operating the Bay of Biscay. Thus, alternative approaches to include their population dynamics in the simulation framework should be implemented. Alternative approaches to low data-demand assessment models, such as the utilization of generic conditioning based on life-history traits of similar stocks or species as used in Fischer et al. (2022b), could be further explored.



Chapter 5

General discussion and future work

5.1 General discussion

This thesis was motivated by the need of ensuring sustainable exploitation of the stocks caught by the demersal mixed-fisheries in the Bay of Biscay. These fisheries catch more than 150 species (Altuna-Etxabe et al., 2020; Briton et al., 2020). However, only approximately one-fourth of these species are included in the TAC and quota system (Briton et al., 2020), while just seven of them and sea bass are included in the Western Waters and adjacent waters multiannual management plan (EU, 2019b). The rest of the stocks are caught without any quota restriction. This raises the question of whether the multiannual management plan of the demersal mixed-fisheries operating in the Bay of Biscay ensures the sustainability of all the species caught.

The evaluation of the existing demersal mixed-fisheries multiannual management plan in the Bay of Biscay was carried out in 2015 by the STECF (STECF, 2015) and resulted in several scientific papers (Bellanger et al., 2018; Prellezo et al., 2016). Two simulation models were implemented: one focused on the French fleets and another on the Spanish fleets. The simulation model for the French fleets included 17 species, whereas the one for the Spanish fleets included 12 species. However, the population dynamics of only three of them (hake, sole and nephrops) in the former and two of them (hake and megrim) in the latter were simulated dynamically. The other species were included in the simulation algorithm using a constant CPUE approach, i.e., their population dynamics were not considered, and it was assumed that their productivity was completely independent of the intensity of the fishing activity. In this thesis, we further developed that simulation work. First, the stocks that should be included in the simulation algorithm to provide a more holistic evaluation of the management plan were identified. The stock selection was based on a species prioritisation approach that allows ranking species from those most at risk, most exploited and/or economically most important, to the least (Chapter 2) (Altuna-Etxabe et al., 2020). The data availability of most of the stocks selected was not enough to apply a conventional stock assessment model to infer their population dynamics and include them in the simulation algorithm. Thus, second, the performance of a stock assessment model appropriate for data-limited situations, namely the Stock Reduction Analysis (SRA) (Huynh et al., 2020a), was evaluated by “self-test” simulation to identify data requirements that ensured an accurate enough model fit (Chapter 3). Last, in Chapter 4, the performance of the current and alternative management strategies, considering the implementation or non-implementation of the landing obligation, was evaluated for the demersal mixed-fisheries in the Bay of Biscay within the simulation framework. The simulation was conducted using FLBEIA model (Bio-Economic Impact Assessment in FLR) (García et al., 2017). The conditioning of data-rich stocks was based on their latest assessment, while the conditioning of four data-limited stocks was based on the SRA model. As a result, the number of stocks included in the simulation framework was extended to 28 stocks, where 13 out of them were included in the simulation using an age-structured population dynamics, and the rest 15 stocks were included using a constant CPUE approach. In addition, both the French and Spanish fleets were included in the algorithm using the same fleet segmentation as used in the provision of ICES mixed-fisheries considerations for the Bay of Biscay (ICES, 2022b,e).

5.1.1 The selection of stocks

The first challenge addressed in this thesis was the selection of the stocks that should be included in the mixed-fisheries simulation algorithm. Although it is practically impossible to include in the simulation framework the large number of stocks caught by a mixed-fishery due to the significant workload, computational resources needed and limitations in available data, the number of stocks should be large enough to describe the activity of the fleet realistically and ensure that the impact of the management plan can be assessed adequately (Ulrich et al., 2011).

The proposed stock selection method was based on the species prioritisation approach developed by Osio et al. (2015). They put forward an approach to prioritise species for data collection and research according to their landing and economic importance as well as their potential risk for the effects of fishing. The former was measured in terms of landing and average price per kilogram, while the later was derived from a risk assessment model. Given that the discards in the demersal mixed-fisheries operating in the Bay of Biscay are high and this can limit the activity of the fishery under the landing obligation (ICES, 2022b; Prellezo and Villasante, 2023), the approach was modified to account also for discards. Therefore, in this thesis, the species were ranked from the most exploited, economically most important and/or most at risk for the effects of fishing, to the least (Altuna-Etxabe et al., 2020). We recommend including species that are ranked at the top, ensuring that they represent a big proportion of the fishery's total catch and revenue, and that the risk associated with the remaining species is not likely to be enhanced by the activity of the fleet. The top ten ranked species represented more than 76% of the total Basque demersal income, landings and discards volume proportion. However, in addition to these 10 species, it was also necessary to include pollack, thornback ray and cuckoo ray in the simulation model. The need to include this species in the simulation framework is because their catch quotas have a high probability of constraining the fishing effort of the fleet under the implementation of the landing obligation. Comparing our simulation study to the one conducted by STECF in 2015, we expanded the scope by including all the stocks in the TAC and quota system that have the potential to restrict the fishing activity of the demersal mixed-fisheries operating in the Bay of Biscay. Furthermore, unlike the 2015 study, our simulation algorithm incorporates the population dynamics of stocks that required additional management measures for their protection, specifically rays and red mullet. These stocks were identified as the ones that need additional management measure for their protection from the simulation model.

The high number of stocks caught by mixed-fisheries and the low data availability of most of the cases, make the use of fully quantitative risk assessment models difficult. Therefore, the risk for the effects of fishing was calculated using a semi-quantitative risk assessment model: the productivity-susceptibility analysis (PSA) (Hobday et al., 2011). The PSA estimates the potential risk of stocks for the effects of fishing activity by considering the biological productivity of the stock and the potential impact of the fishing activities on the stock (Hobday et al., 2007). The cumulative impact of the various gears used by the fisheries was accounted for by using the aggregated susceptibility proposed by Micheli et al. (2014). Although the results of the PSA from individual assessments should be interpreted with caution, the relative comparisons of potential risk among stocks are considered robust (Micheli et al., 2014). In our study, sharks and rays had the highest risk values, positioning them at the top of the risk ranking and emphasizing the potential necessity for their protection. Conversely, the majority of pelagic stocks were ranked at the bottom of the risk ranking, indicating the minimal impact of alternative management strategies employed by the demersal mixed-fisheries on their sustainability. The results obtained from the PSA were confirmed by the simulation results. The simulation model results confirmed the need to provide protection to rays, as well as the low risk of pelagic stocks being negatively affected by the impacts of the fishery. Thus, the value of using PSA as a first quick approach to assess the relative potential risk of stocks for the effects of fishing activity was underscored as demonstrated by previous studies (Hobday et al., 2007; ICES, 2012; MSC, 2001; Ormseth and Spencer, 2011).

Unlike other simulation studies that include stocks based exclusively on landing volume or economic value (García et al., 2019; Hamon et al., 2007; Prellezo et al., 2016), our species prioritisation approach also included an ecological factor to identify the stocks that should be considered in the simulation. We found that the ecological factor is a relevant variable when choosing the stocks to include in the simulation framework. In fact, considering the ecological factor, specifically the potential risk of stocks for the effect of fishing, we identified some stocks as important to include in the simulation due to their high risk for the effects of fishing that otherwise would not have been identified. For example, rays, which had a high probability of having their SSB below the limit biomass reference points according to the simulation study results, would not have been identified as important stocks to include in the simulation if only landings volume and/or economic aspects were considered. This is because their contribution to the landings and revenue of the fleet was low. Therefore, this study has confirmed the importance of a balanced contribution of the economic and the ecologic sustainability factors in the ranking to select the stocks to be included in the simulation. The resulting list could be used not only for selecting stocks to be included in an simulation algorithm for mixed-fisheries, but also to identify data collection and research priorities (Osio et al., 2015).

5.1.2 The "self-test" simulation study of the stock reduction analysis

A semi-quantitative analysis, such as PSA, is a useful tool to have a general overview of the potential risk of the stocks caught (Hobday et al., 2007; Patrick et al., 2010). However, it is not sufficient to condition stocks population dynamics in the MSE algorithm (García et al., 2017). Data-rich stocks usually have a quantitative assessment in which the conditioning of their population dynamics in the simulation can be based. However, most of the global fish resources lack a quantitative assessment (Costello et al., 2012) and their data availability is not enough to assess their population dynamics with conventional assessment models (Carruthers et al., 2014). Therefore, the conditioning of data-limited stocks in the MSE has to be based on life-history traits and additional assumptions on the exploitation pattern (e.g., Carruthers et al., 2014; Fischer et al., 2020) or on low data demand assessment models (e.g., Geromont and Butterworth, 2014; Mildenberger et al., 2022).

The SRA model is a low data demand age-structured stock assessment model compared to conventional age-structured stock assessment models such as, assessment for all (a4a) by Jardim (2017) or state-space assessment model (SAM) by Nielsen and Berg (2014). The SRA model has been proposed as alternative to conventional age-structured assessment models for situations where catch-at-age data are not available (Huynh et al., 2020a; Kimura and Tagart, 1982; Walters et al., 2006). It has been used in several studies to define the population dynamics of data-limited stocks in a MSE framework (Hordyk et al., 2019; Huynh et al., 2020b). Biomass-dynamics models are usually used in data-limited situations as they only need catch data and abundance index data (Abaunza et al., 2003). In contrast to biomass-dynamics models, the SRA model also needs life-history parameters (e.g., individual growth, recruitment, maturity etc.) and information on fleet and index selectivities. The life-history parameters are the primary data source known for most of the data-limited stocks, or in the worst case, they can be borrowed from similar stocks and/or species (Bentley, 2014; Punt et al., 2011), whereas the selectivity parameters can be derived from length-structured data. Beyond these minimal data requirements, the SRA model is flexible to use additional data from other multiple sources (e.g., annual mean length in the catch, catch-at-length, catch-at-age and index-at-age data) and to handle scarce and scattered data (e.g., covering discontinuous and short time periods). Moreover, it allows to include uncertainty in all the life-history parameters and in catch and abundance index data. This is an interesting feature from the perspective of the simulation model conditioning as it provides a common framework for the integration of the main uncertainties in the stock dynamics.

To evaluate in which cases the application of the SRA model provides accurate enough estimates, we conducted a "self-test" exercise which complemented the work done by Hordyk et al. (2019). While Hordyk et al. (2019) evaluated the performance of the SRA model under bias in biological parameters and error in observation and model assumptions, in this thesis we evaluated the performance of the SRA model under alternative data availability scenarios (depending on the lengths and structures of the time-series of catch and abundance index data), different population exploitation levels, misspecification in fishery selectivity and initial population assumptions. We made progress in understanding how the SRA model works and under which data availability cases it can be implemented reliably to estimate data-limited stock population dynamics. In addition, to the extent of our knowledge, the effect of alternative fished equilibrium assumption as an initial population proxy on the performance of an assessment model has never been evaluated. The fished equilibrium assumption is used as an initial population proxy in other stock assessment models, for example in the widely used Stock Synthesis (Methot and Wetzel, 2013). We found that it can have a high impact on the model results. When the available data started from an unfished condition, time-series of total catch and abundance index that covered one stock generation time were sufficient to obtain accurate population and exploitation level estimates. However, in most of the cases, it is not likely to know the data from the beginning of the exploitation state. In Europe there is a long tradition of fishing and most of the stocks have been fished for many decades or even centuries, with industrial fishing developing after the World War II. However, the data collection framework program did not start until 2000. At its beginning, the data collection regulation was focused on the sampling of target species, regulated under Regulation (EC) No 199/2005. However, it was not until 2008 that a change occurred in the data collection framework for all regulated fisheries, with the introduction of Regulation (EC) No 199/2008, which shifted the focus towards the collection of data for all the species caught. According to the results from the "self-test" exercise, when the available data was not long enough to cover the whole exploitation period, the time-series of total catch and abundance index had to cover at least two stock generation times to obtain accurate estimates from the SRA model. When the length of the data time-series was shorter than two stock generation times, additional data covering one stock's generation time was needed. Mean length in the catch data was sufficient to obtain accurate relative estimates (e.g., depletion). However, for accurate estimates of unfished recruitment, age- or length-structured data were necessary.

The results of the “self-test” simulation study also provided valuable information to data collection strategies in data-limited situations. Assessing the value of each data-source in an assessment model is important for identifying the most cost-efficient way to improve the accuracy of population and exploitation level estimates, which would in turn result in better management of the stock (Magnusson and Hilborn, 2007). Each data source provides different types of information, and the cost of collecting each dataset can vary substantially (Begg et al., 2005; Dennis et al., 2015). Fishery data are generally less expensive to collect compared to fishery-independent data. However, fishery-independent data allow you to better understand population dynamics than fishery data. Usually, the collection cost of age-structured data is higher than the collection cost of length-structured data. On the one hand, we quantified the value of fishery age- and length-structured data to obtain accurate estimates from the assessment model. We found that when the stock’s somatic growth variability was low and the life-history parameters were known, the use of length-structured or age-structured data gave similar estimates. In such cases, the additional cost of obtaining age-structured data is unnecessary, and length-structured data is sufficient for accurate population dynamics estimates from the assessment model. On the other hand, we quantified the value of fishery and fishery-independent data to obtain accurate estimates from the assessment model. We found that when the fishery targeted small individuals and selectivity was estimated from age- or length-structured fishery data, without fishery-independent age-structured population abundance index data, the abundance of the oldest individuals was overestimated. In fact, in the absence of age-structured population data, the model assumed that old individuals not caught by the fishery experienced only natural mortality. In these cases where the selectivity was focus on small individuals and was calculated from age- or length-structured fishery data, the inclusion of age-structured fishery-independent data was necessary to provide information about the abundance of the oldest individuals. Data-limited stocks typically receive limited research time and investment for data collection and analysis (Bentley, 2014), making difficult to obtain fishery-independent data. In these cases (i.e. in the absence of age-structured fishery-independent data when the selectivity was estimated from fishery age-structured data, and the fishery primarily caught small individuals), our “self-test” simulation study revealed that to fix the descending limit of the selectivity curve at a value higher than 0.25 could be appropriate to obtain accurate estimates from the assessment model.

Punt et al. (2020) identified simulation testing as one of the essential features of an assessment model. This allows, on the one hand, to evaluate the estimation performance of the assessment model by “self-test” simulation study and, on the other hand, to use the assessment model as part of the components of the MSE (either for conditioning the operating model or as the assessment model that informs the management procedure). Both aspects have been well covered in this thesis. The code required for “self-test” simulation study of the SRA model was developed in R as part of the thesis (Chapter 3) and the SRA model results were used for conditioning some of the selected data-limited stocks in the MSE as described in Appendix F.

5.1.3 Conditioning of data-limited stocks

The population dynamics of four data-limited stocks, catshark, red mullet, thornback ray and cuckoo ray, in the simulation model were conditioned using the SRA model. The catch and index data time-series of these stocks started from an exploited condition and covered less than two generation times but more than one generation time of the stocks (ICES, 2020a,b,d, 2021e; Ifremer, 2020). In addition, catch-at-length data were available for more than one stock generation time. According to the "self-test" simulation study (Chapter 3), these data were enough to obtain reliable population dynamics estimates from the SRA model.

All the life-history parameters were taken from the literature, and for the first year of the time-series, the fished equilibrium condition was assumed to be equal to the mean of the first five years' catches, as recommended in Chapter 3. Efforts were made to incorporate a high level of uncertainty (i.e., a coefficient of variation of 25%) using a uniform distribution around the mean parameter values for all the data and life-history parameters in order to account for the great variability in the data-limited stocks population dynamics. However, these attempts resulted in implausible population dynamics estimates, including extremely high R_0 values. This could be related to the fact that correlation between the values in life-history parameters was not considered when introducing uncertainty. As a results, the incorporated uncertainty was limited to obtain sensible results. Different uncertainty levels were incorporated for the conditioning of the fished equilibrium assumption, the catch and abundance index data and the life-history parameters, with the aim of covering the real value and existing uncertainty. A high coefficient of variation was included for the fished equilibrium assumption (CV equal to 25%), while a lower coefficient of variation was used for catch and abundance index data and for more uncertain life-history parameters, such as natural mortality and recruitment parameters (CV equal to 10%). Even lower coefficients of variation were applied to life-history parameters that were considered less uncertain, such as length-weight relationship parameters, growth parameters and maturity parameters (CV equal to 5%). The uncertainty included resulted in large confidence intervals for the population dynamics and exploitation level estimates. However, it is important to note that the uncertainty included in the simulation model was not based on real data, and therefore, the output uncertainty may not accurately represent the true uncertainty in the system.

5.1.4 Management strategy evaluation

The developments made on the implementation of the MSE regarding the evaluation of multiannual management plan of mixed-fisheries operating in the Bay of Biscay constituted a significant improvement in comparison to previous work (STECF, 2015). The operating model included the age-structured population dynamics of more stocks than in the previous work and an updated characterization of the fleets. Compared to the simulation of the STECF, the population dynamics of another 6 data-rich stocks (black anglerfish, white anglerfish, horse mackerel, mackerel, sea bass and blue whiting) and 4 data-limited stocks (catshark, red mullet, thornback ray and cuckoo ray) were also included in the simulation. The population dynamics of Norway lobster was included in 2015 by STECF however it was not included in this simulation study. Since 2016 its assessment uses an underwater TV survey to obtain an absolute value of its biomass (ICES, 2022d), and the assessment model results are not translatable to a classical population dynamics model. Thus, in this thesis, a constant CPUE was used to include Norway lobster in the simulation.

We found that the sustainability of all data-rich stocks included in the simulation was ensured with the management of few data-rich stocks included in the Western Waters and adjacent waters multiannual management plan (EU, 2019b). However, the sustainability of all data-limited stocks was not ensured with the current management plan. The current management plan based on the management of few data-rich stocks, only ensured the sustainability of one of the four data-limited stocks included in the simulation, namely, catshark. The SSB of red mullet, thornback ray and cuckoo ray had a high probability of being below the limit reference biomass in the long-term, regardless of whether the landing obligation was in place or not. Thus, additional management measures are needed to ensure the sustainability of the system. Although it is true that the uncertainty included to condition data-limited stocks gave very large confidence intervals for the SSB, the results highlighted the lack of robustness of the current management strategy to the existing uncertainty.

In this thesis, the effects of alternative management strategies on the sustainability of the system were also evaluated in order to identify how the sustainability of the system can be improved with respect to the current management plan. We found that the sustainability of red mullet improved when it was included in the TAC and quota system and its catch advice was generated based on an abundance index or using the MSY advice rule. Additionally, the sustainability of thornback ray improved when the advice of it was given using the MSY advice rule instead of the advice rule based on an abundance index. The sustainability of cuckoo ray did not improve with alternative management strategies tested in this study. This lack of improvement was attributed to its low initial biomass level. The probability of cuckoo ray's SSB being below the limit reference biomass exceeded 50% in the first year of the simulation. Demersal stocks, including cuckoo ray, exhibit characteristics such as low productivity, long lifespan and late maturity, which diminish their capacity to recover from low biomass levels. Therefore, the recovery of this particular stock requires a more customized and specific management strategy that extends beyond the scope of the analyses conducted in this thesis. For instance, analysing the spatio-temporal distribution of cuckoo ray population and fisheries would help identify overlap areas and establish targeted management measures, such as specific spatio-temporal fishery closures.

Rays and anglerfishes are managed by grouped TACs at genus level due to the difficulty to distinguish between stocks of the same genus. In the Bay of Biscay, the performance of grouped TACs has never been evaluated before and the existing evaluation of management strategies has been done at stock level (García et al., 2019). This is also the case in the ICES mixed-fisheries considerations (ICES, 2022b). In this thesis, we found that while the grouped TACs of rays and anglerfishes promoted the stability of the fishing activity, it could compromise the sustainability of the stocks. When individual TACs for rays and anglerfishes were used, the TAC of thornback ray limited the fishing activity of most of the fleets and produced a misuse of other stocks fishing opportunities that was alleviated when the grouped TAC was used. However, the use of grouped TACs resulted in fishing levels higher than the recommended at single stock level, and it had a negative impact on the SSBs of rays and anglerfishes.

This thesis has contributed to the evaluation of the management of the mixed-fisheries operating in the Bay of Biscay, and has helped moving forward towards one of the main objectives of the CFP which is to ensure the sustainability of all the stocks caught while maintaining the fishing activity in the long-term.

5.2 Future work

The selection of the stocks to include in the simulation was done based on the risk for the effects of fishing activity calculated by PSA, the catch level, and the economic importance. However, the method could be broadened to include other factors relevant for the case study, such as potential choking effects. In addition, spatial aspects could be incorporated to the ranking by calculating a spatially explicit risk of the species for the effects of fishing. The standard PSA accounts for horizontal and vertical overlap between fleet and species distribution for the calculation of the susceptibility. However, neither effort nor species distribution are homogeneously distributed along the whole area. Thus, calculating the spatial potential risk of the species for the effects of fishing activity as done by Roberson et al. (2022) in the Indian Ocean would suppose an improvement in comparison to the current ranking.

The "self-test" simulation served to evaluate the SRA model's performance depending on the length and structure of the time-series of catch and abundance index data, alternative population exploitation levels, different initial population assumptions and misspecification in fishery selectivity. However, other parameters like survey selectivity and biological parameters were assumed to be known exactly. Additional simulations incorporating misspecification across survey selectivity and biological parameters would provide a complete evaluation of the SRA model's capability to estimate population dynamics accurately under data-limited situation.

Despite the significant improvement on the number of stocks included in the simulation, the dynamics of 15 stocks were still modelled using the constant CPUE approach. Alternative methods to mimic their population dynamics, such as generic conditioning based on life-history traits of similar stocks or species (Fischer et al., 2022b; Jardim et al., 2015), should be investigated. Particularly, the research effort should be focused on the conditioning of pollack and whiting population dynamics because they are in the TAC and quota system, and their catch advice had high probability to limit the fishing effort of most of the demersal mixed-fisheries operating the Bay of Biscay.

More realistic fleet dynamics models should be used to improve the predictive capability of the model. On the one hand, the possibility to extend the model to incorporate explicitly additional fleets to account for the catches of the stocks out of the Bay of Biscay should be considered in any future work. Adding these fleets could lead to the need of incorporating more stocks (i.e., cod or haddock) and eventually to merge the Bay of Biscay and Celtic Sea mixed-fisheries models (ICES, 2022b,f). On the other hand, in FLBEIA there is a profit maximization approach that calculates the total effort and the effort allocation among metiers that maximizes the profit (García et al., 2017). In this study, economic performance statistics such as market's impact (i.e., prices by fish size) and structural cost (i.e., labour, fuel and oil and other material costs) were not considered, however, they can drive the fleet dynamics (Marchal and Vermard, 2022). Thus, in further analysis it would be interesting to implement a profit maximization approach in FLBEIA and evaluate if it improves the predictive capability of the model doing a hindcasting in past data. Marchal et al. (2013) found that for the French trawlers harvesting demersal stocks, the fleet dynamics was driven 80% by tradition and 20% by profit maximization. Currently it is not possible to mix tradition and profit maximization in FLBEIA and a new approach should be developed in this kind of approach.

Due to time restrictions some management scenarios were not evaluated in the simulation and should be further considered in any future work. These include the exemptions to the landing obligation (the exemption due to the high survivability for rays and the minimis exemptions for hake, megrim, anglerfishes, sole, mackerel and horse mackerel), the use of fishing mortality ranges around F_{MSY} to mitigate the choke effect of the stocks, and the new HCRs for data-limited stocks proposed recently by ICES (2022a).

Finally, despite the uncertainty sources already included in the simulation, it is acknowledged that two major sources of uncertainty (Punt et al., 2016) were underestimated. On the one hand, no observation error and assessment error were included. On the other hand, biomass and fishing mortality reference points were assumed to be known. Including them could affect the risk estimates and their inclusion may deserve further attention.

Chapter 6

Conclusions and thesis

This chapter lists the main conclusions in relation to each of the three specific objectives and evaluates the validity of the working hypothesis stated in the introduction section (Chapter 1).

The first objective of this thesis was: "To identify the most at risk, most exploited and/or economically most important stocks for the fishery that should be included in the management strategy evaluation framework" (Chapter 2). The main conclusions in relation to this objective are:

1. The proposed species prioritisation approach allowed ranking species from those most at risk, most exploited and/or economically most important, to the least. The resulting ranking was useful to identify the stocks that should be included in the management strategy evaluation framework. The results could also be used to identify data collection and research priorities.

2. The contribution to the ranking of each of the three factors (risk, catch, revenue) was balanced. So, in contrast to other stock selection methods, the ecologic factor was as important as the economic factors in the process of identifying the stocks that should be included in the simulation.

3. The productivity-susceptibility analysis (PSA) was a useful semi-quantitative risk assessment model to assess within the same framework the ecological sustainability of the large number of stocks involved in a mixed-fishery despite the differences in data availability.

The second objective was: "To evaluate by "self-test" simulation the ability of the SRA model to obtain accurate population dynamics estimates under alternative data availability scenarios" (Chapter 3). The main conclusions in relation to this objective are:

4. As expected, the more data and longer time-series used, the higher the accuracy of the SRA model estimates.

5. When the available data started from an unfished condition, time-series of total catch and abundance index that covered one stock generation time were enough to obtain accurate population and exploitation level estimates.

6. When the available data started from an exploited condition, the time-series of total catch and abundance index had to cover at least two stock generation times to obtain accurate estimates. When the length of the data time-series was shorter than two stock generation times, additional data covering one stock's generation time was needed. Mean length in the catch data was sufficient to obtain accurate relative estimates, such as depletion. However, for accurate estimates of unfished recruitment, age- or length-structured data were necessary.

7. When the variability in the stock's somatic growth was low (CV equal to 0.15) and the biological parameters were known, the use of length-structured data or age-structured data gave similar estimates.

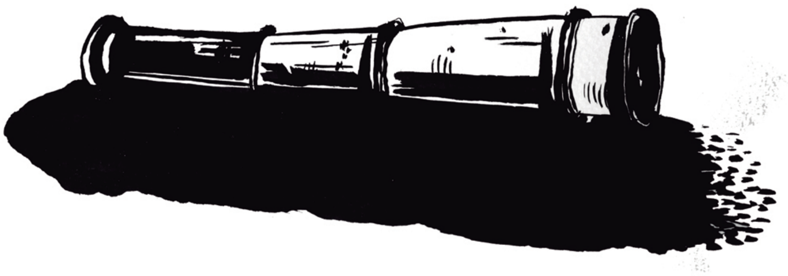
8. The addition of one stock generation time of index-at-age data and age- or length-structured catch data to total catch and abundance index data was the best option to obtain accurate estimates, especially when the fishery selectivity was unknown.

The third objective was: "To evaluate the impact of current and alternative management strategies for the demersal mixed-fisheries operating in the Bay of Biscay on the sustainability of the most emblematic stocks" (Chapter 4). The main conclusions in relation to this objective are:

9. The conditioning of the simulation model constituted an improvement in comparison to previous works as the operating model included the age-structured population dynamics of more stocks. Apart from the population dynamics of hake, sole and megrim, the population dynamics of another 6 data-rich stocks (black anglerfish, white anglerfish, horse mackerel, mackerel, sea bass and blue whiting) and 4 data-limited stocks (catshark, red mullet, thornback ray and cuckoo ray) were incorporated. In addition, 51 fleets divided into 79 metiers were included in the algorithm with the same segmentation as used in the provision of ICES mixed-fisheries considerations for the Bay of Biscay.
10. The current management plan ensured the sustainability of all the data-rich stocks and one of the four data-limited stocks included in the simulation, namely, catshark. On the contrary, the SSB of red mullet, thornback ray and cuckoo ray had a high probability of being below the limit reference biomass in the long-term, regardless of whether the landing obligation was in place or not.
11. The sustainability of red mullet and thornback ray improved substantially with some changes in their management. Specifically, when red mullet was included in the TAC and quota system based on empirical or model-based HCRs, and when the HCR of thornback ray was upgraded from the empirical to the model-based HCR. The status of cuckoo ray did not improve with any of the alternative management strategies tested.
12. Grouped TACs for rays and anglerfishes promoted the stability of the fishing activity, but compromised the sustainability of the individual stocks.
13. Stock-recruitment relationship parameters were the factors that had the highest impact on the performance of the management strategies. Additionally, the implementation of the landing obligation also had a high impact on the performance of the management strategies, and the type of HCR contributed highly to explain the variability of the last year SSB.

Considering all these conclusions, the working hypothesis has been validated, resulting in the following thesis:

An adequate stock selection based on both economic and ecologic factors and a reliable representation of data-limited stock dynamics using a low data demand assessment model, allowed to carry out a holistic evaluation of the multiannual management plan for the demersal mixed-fisheries operating in the Bay of Biscay. The evaluation revealed that the current management plan ensures the sustainability of the data-rich stocks and the data-limited catshark stock, but is insufficient for the other three data-limited stocks (red mullet, thornback ray and cuckoo ray) included dynamically in the simulation.



Chapter 7

Scientific production and formation

The scientific and technical material produced, and the scientific formation acquired during the period of this doctoral thesis are detailed in this section:

Scientific articles

Altuna-Etxabe, M., García, D., and Ibaibarriaga, L. Multiannual management strategy evaluation of the demersal mixed-fisheries operating in the Bay of Biscay. In preparation.

Altuna-Etxabe, M., García, D., Ibaibarriaga, L., Huynh, Q.C., Murua, H., and Carruthers, T.R. The value of data in stock assessment models with misspecified initial population and selectivity. In preparation.

Altuna-Etxabe, M., Ibaibarriaga, L., García, D., and Murua, H. 2020. Species prioritisation for the development of multiannual management plans for the Basque demersal fishery. Ocean & Coastal Management, 185:105054.

Scientific reports

ICES, 2022. Working Group on Mixed Fisheries Methodology (WGMIXFISH- METHODS). ICES Scientific Reports. 4:60. 100 pp. `http://doi.org/10.17895/ices.pub.20401389`

Conferences

Miren Altuna-Etxabe, Dorleta García, and Leire Ibaibarriaga (2022). Evaluation of the demersal mixed fisheries multiannual management plan in the Bay of Biscay including non-target and bycatch stocks (ICES ASC 2022). Oral communication. Hybrid conference. 19th-22nd September 2022.

Miren Altuna-Etxabe, Dorleta García, Leire Ibaibarriaga, and Hilario Murua (2022). Multiannual management plans evaluation for the Basque demersal fishery (RiMer 2022). Oral communication. San Sebastian 4th February 2022.

Miren Altuna-Etxabe, Dorleta García, Leire Ibaibarriaga, and Hilario Murua (2021). The value of assessment data to obtain accurate stock status indicators (ICES ASC 2021). Oral communication. Virtual conference. 6th-10th September 2021.

Miren Altuna-Etxabe, Dorleta García, Leire Ibaibarriaga, and Hilario Murua (2019). Good practice for conditioning data-limited stocks in management strategy evaluation framework (ICES ASC 2019). Oral communication. Gothenburg (Sweden) 9th-12th September 2019.

Miren Altuna-Etxabe, Dorleta García, Leire Ibaibarriaga, and Hilario Murua (2019). Multiannual management plans evaluation for the Basque demersal fishery (RiMer 2020). Oral communication. San Sebastian 7th February 2020.

Miren Altuna-Etxabe, Leire Ibaibarriaga, Dorleta García and Hilario Murua (2018). Productivity and Susceptibility Analysis (PSA) of the species caught by the Basque demersal fleets in the Bay of Biscay. XVIth International Symposium on Oceanography of the Bay of Biscay (ISOBAY 16). Oral communication. Anglet (France) 5th-7th June 2018.

Workshops

ICES working group on mixed fisheries advice methodology (WGMIXFISH- METHODS), in IFREMER, Nantes (France). 20^{th}-24^{th} June 2022.

ICES working group on data-limited stocks of short-lived species (WKDLSSLS), in AZTI, Pasaia (Spain).16^{th}-20^{th} September 2019.

ICES working group on mixed fisheries advice methodology (WGMIXFISH- METHODS), in IFREMER, Nantes (France). 15^{th}-19^{th} October 2018.

ICES Working Group on Multispecies Assessment Methods (WGSAM), in AZTI, Pasaia (Spain). 16^{th}-20^{th} October 2017.

Research projects

SEAWISE: Shaping Ecosystem Based Fisheries Management (2021-2025). Funded by the European Commission.

NEXTSGP-GESTACOMNX: Management of non-target and target species in the demersal mixed-fishery operating in the national fishing ground (2022). Funded by the Spanish government.

PROBIFISH: Protecting bycaught species in mixed fisheries (2018-2021). Funded by the European Commission.

IMPACPES: Development of tools to assess the biological and economic impact of fisheries management (2018-2019). Funded by the Basque Government.

Research stays

Simulation and evaluation of alternative multiannual fishery management plans. Marine Institute (MI) in the Fisheries Ecosystems Advisory Services (FEAS) department with Prof. David Reid. Galway (Ireland) 1^{st} September- 1^{st} December.

Data-limited species conditioning using DLMToolkit (Data-Limited Methods Toolkit). University of British Columbia (UBC) with Dr. Thomas Carruthers. Vancouver (Canada) 26^{th} March-10^{th} July 2019.

Courses

Ecological Forecasting. Oslo (Norway) 10^{th}-15^{th} October 2019.

MSC-DLMToolkit. Data Limited Methods Project (MSC-DLM). Cadiz (Spain) 9^{th}-10^{th} October 2018.

FIBEIA Training course: Bio-Economic lmpact Assessment using FLR. Delivered by AZTI and imparted by Dorleta García, Sonia Sanchez and Agurtzane Urtizberea. Derio (Spain) 3^{rd}-5^{th} October 2017.

Appendix A

General Introduction

In the documentary entitled *We are nature*, Jane Goodall says "we may think that nature is something separate and we can distance ourselves and live in our little bubble, but it's not true". Human health, both in physical and mental terms, is related to the biodiversity of the ecosystems (Naeem et al., 2016). The biodiversity provides critical services to the well-being of the human society (Naeem et al., 2016). It provides, among others, food and water, clean energy, pharmaceutical and cosmetic products, coastal defence and regulation of climate, without detracting that indirectly it plays an important role in economic activities (Barbier, 2017; Hanley et al., 2015). All species have a role to play in the ecosystem, and the extinction of even a small and seemingly insignificant species can have a ripple effect on other species leading to ecosystem collapse and consequently, endangering the human health (Naeem et al., 2016).

It is considered that we are in the sixth and potentially most extreme biodiversity crisis, where a large number of species are predicted to extinct at a faster rate than ever before (Banks-Leite et al., 2020). Oceans are not an exception and marine ecosystems are disappearing at a rapid rate and on an alarming scale worldwide (Barbier, 2017). There is a widespread degradation in marine ecosystems and biodiversity due to the cumulative effects of human activities (McQuatters-Gollop et al., 2022). Some of the main threats are related to direct human pressures such as coastal habitat destruction, pollution, non-local species introduction and fishing (McQuatters-Gollop et al., 2022). In addition, there are other indirect effects of human activity like climate change (McQuatters-Gollop et al., 2022). The biodiversity loss is exacerbated by the expansion and growth of human population, which has a positive relationship with the decline of taxonomic groups and ecosystem structures (Lotze et al., 2011). Nowadays, the ability of the ocean to support human demand is at a crossroads (Duarte et al., 2020).

Until the end of the 19^{th} century, it was a general believe that the oceans were so vast that fishing practices were unable to impact negatively on fish population abundance. Thomas Henry Huxley, in the inaugural presentation of the International Fisheries Exhibition in 1883 said "the cod fishery, the herring fishery, the pilchard fishery, the mackerel fishery and probably all the great sea fisheries, are inexhaustible; that is to say, that nothing we do seriously affects the number of the fish. And any attempt to regulate these fisheries seems consequently, from the nature of the case, to be useless" (Huxley, 2020). However, before the end of the 19^{th} century, the decrease of fish abundance in several fishing grounds began to raise voices against the thought of the inexhaustibility of fish stocks and against the thought of the lack of destructive power of human activity on the ocean resources (Sims and Southward, 2006). Nowadays, it is well known that overfishing is one of the main causes of the decline of marine resources (Barbier, 2017).

According to the Food and Agriculture Organization (FAO) of the United Nations, the number of overfished marine stocks has increased in the last decades (Figure A.1), reaching 35 percent of the world fishery resources in 2019 (FAO, 2022). Regarding biologically sustainable stocks (i.e., non overfished stocks), the number of stocks fished at its maximum sustainable yield (MSY) level has increased in the last decades from 51% in 1974 to 58% in 2019, whereas the number of underfished stocks has decreased from 39% in 1974 to 6% in 2019 (Figure A.1). In terms of landings, it is estimated that 18% of the landings come from overexploited fish stocks (FAO, 2022).

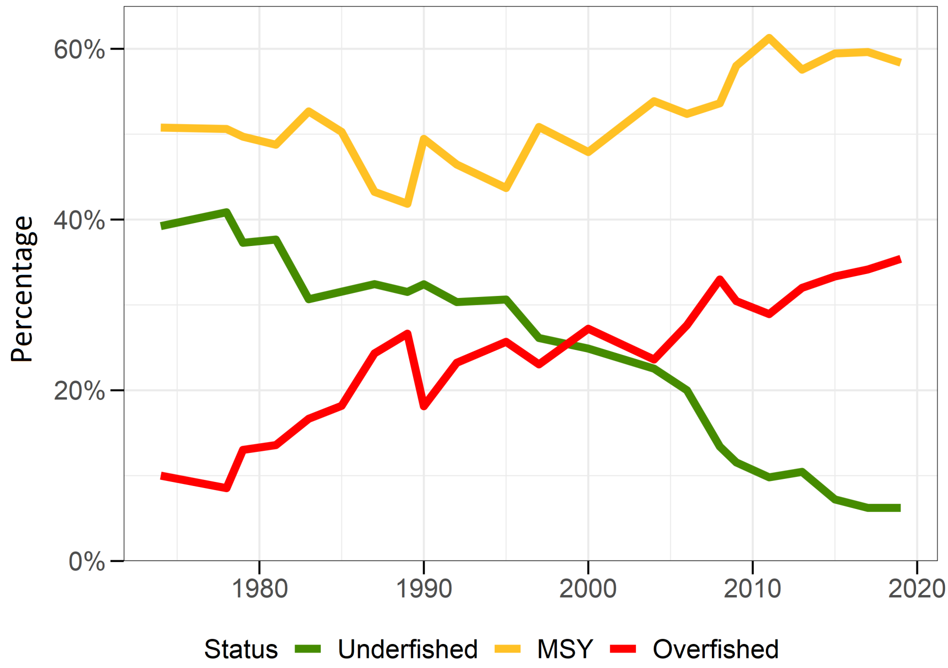


Figure A.1: Global trends in the state of the world's marine fishery stocks from 1974 to 2019. Adapted from FAO (2022).

The percentage of overexploited stocks changes by region (FAO, 2022). In 2019, the highest percentage of overexploited stocks was in the Southeast Pacific (66.7%) followed by the Mediterranean and Black Sea (63.4%) (Figure A.2). In contrast, the Northeast Pacific, Eastern Central Pacific, Western Central Pacific and Southwest Pacific had the lowest proportion of overexploited stocks (between 13% and 23%) (Figure A.2). In other areas, the percentage of overexploited stocks varied between 27% and 45% (Figure A.2).

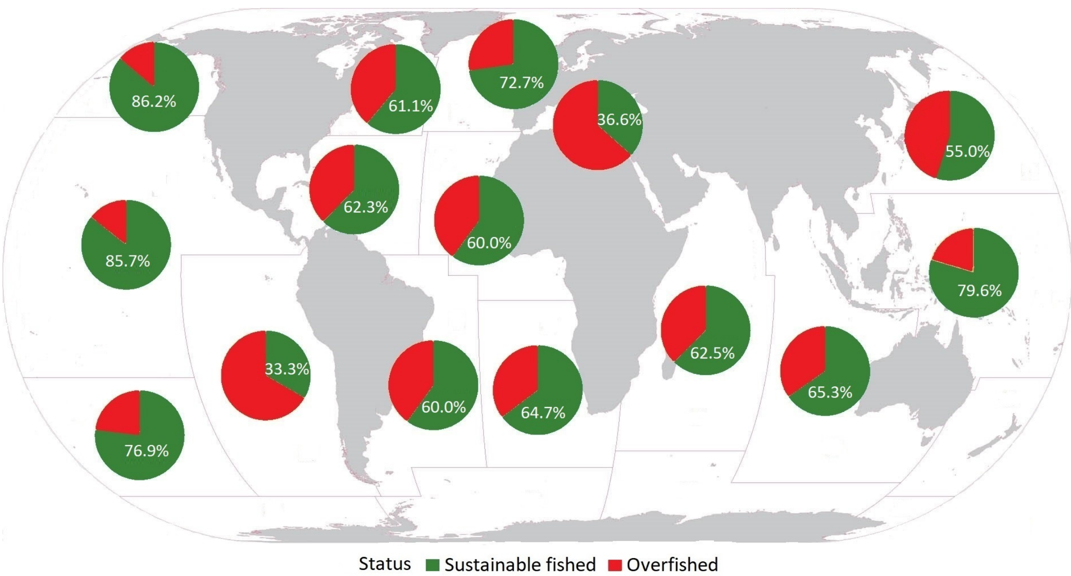


Figure A.2: Percentages of biologically sustainable and overfished stocks by FAO fishing areas in 2019. The numbers represent the proportion of biologically sustainable stocks. Adapted from FAO (2022).

The percentage of overfished stocks in each region is related to the science-based fisheries management intensity in the region (Hilborn et al., 2020). In regions where fisheries are intensively managed, on average, stock abundance is improving or remaining near management target level (Figure A.3). In contrast, in regions with less developed fisheries management, the status of stocks is poorer, and the harvest rates are much greater (Figure A.3). This highlights that science-based fisheries management is crucial to recover or to maintain marine ecosystems in a healthy state (Hilborn et al., 2020).

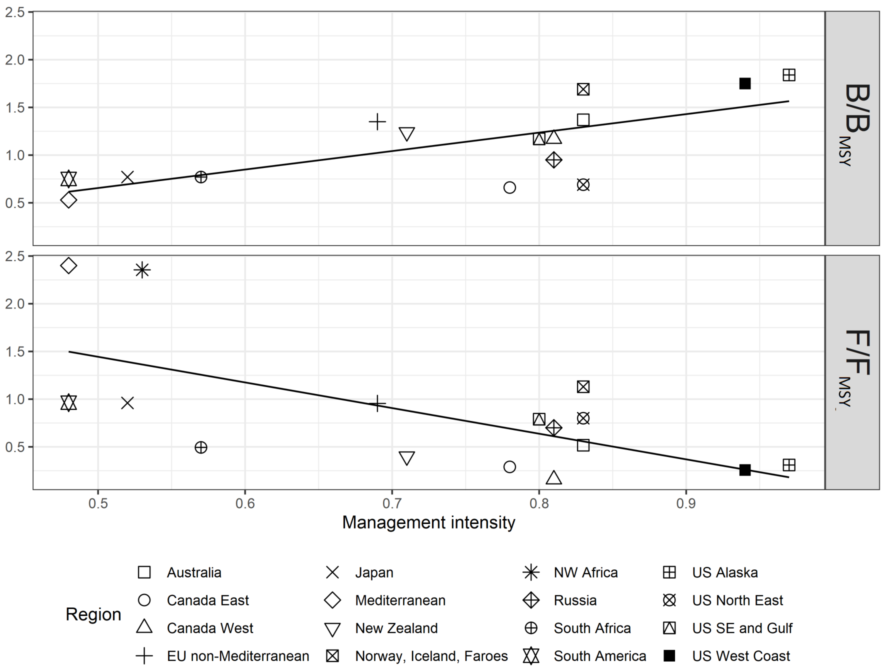


Figure A.3: Regional geometric mean of biomass divided by the MSY biomass (B/B_{MSY}, in the top) and fishing mortality divided by the MSY fishing mortality (F/F_{MSY}, in the bottom) with respect to the management intensity in corresponding regions. Solid line is a linear fit to the data. Adapted from Hilborn et al. (2020).

A.1 Fisheries management in the European Union

A.1.1 Common Fisheries Policy

The Common Fisheries Policy (CFP) is the framework for fisheries management in the European Union. It comprises a set of rules that aim to ensure that fishing and aquaculture are environmentally, economically and socially sustainable and that they provide a source of healthy food for European Union citizens (Casey et al., 2016).

The CFP has its origin in the paper "Basic principles for a Common Fisheries Policy" published in 1967 by the European Commission (Penas, 2016). In 1970 the European Council adopted for the first-time a specific legislation to establish a common organisation of the market for fishery products and started to put in place a structural policy for fisheries (Penas, 2016). At that time, some of the most important fish populations were perceived to be healthy and the development of management methodologies was still limited (Penas, 2016). Thus, the legislation was not focused on the conservation of the resources but on the market of the fishery resources (Penas, 2016).

The birth of the CFP was the adoption in 1983 of Regulation (EEC) No 171/83 and Regulation (EEC) No 170/83. The former laid down technical measures for the conservation of fishery resources such as minimum mesh sizes, bycatch rates, minimum fish landing sizes, and protected areas, while the latter established the general regime for conservation and management of fishery resources including Total Allowable Catches (TACs) as the main management measure (Penas, 2016). The TACs refers to the maximum amount of fish or other marine species that can be harvested from a specific fishery or fishing area during a designated period of time. The TACs are shared as fishing quotas among the Member States of the European Union, and the Member States allocate their fishing quotas among their fleets. Since 1983, the fishing quotas are shared following the relative stability principle, that as its name indicates, means that the TAC of each fish stocks is divided according to a fixed proportion by the Member States of the European Union (Hoefnagel et al., 2015; Penas, 2016). At that time, despite some stocks were managed limiting the amount of fish caught and there were several technical measures, overall the fishing activity followed the principle of free access (Penas, 2016).

Since its beginnings, the CFP is reviewed and reformed every 10 years approximately. The 1992 revision had to address the imbalance between fleet capacity and catch potential after the withdrawal of Greenland in 1985, the accession of Spain and Portugal in 1986 and the reunification of Germany in 1990. In addition, the access of a fishery to a specific water became regulated by a licencing system. However, the measures adopted were not sufficiently effective and the depletion of many fish stocks continued at an even faster rate, leading to the 2002 reform of CFP (Breuer, 2022).

In the 2002 reform of the CFP, the Regulation (EC) No 2371/2002 included for the first-time as overarching goal the sustainable use of fishery resources by guaranteeing stable incomes and jobs for fishers and supplying consumers (Breuer, 2022). The reform specified that sustainability must be based on scientific advice and the precautionary principle. Under the precautionary principle, the lack of full scientific certainty should not be the reason for postponing the implementation of measures that could prevent environmental degradation (UNCED, 1992). Moreover, the reform of the CFP in 2002 introduced the long-term approach to fisheries management, including multiannual recovery plans for stocks outside safe biological limits and multiannual management plans for other stocks to ensure their sustainability. This changed the fundamental sustainability paradigm from tactical decisions to achieve short-term goals to strategic decisions to achieve long-term objectives of multiannual management plans.

The last reform of the CFP, the Regulation (EU) No 1380/2013, was carried out in 2013. The policy-oriented fisheries science was reinforced by intensifying the collection of data, and sharing of information on stocks and fleets (Breuer, 2022). In addition, better cooperation between industry and scientists was fostered and the decision making was regionalized: the European Union legislators draw up the general framework and the Member States develop the implementing measures by cooperating on a regional level (Breuer, 2022). The overriding management objective of the latest CFP reform in 2013 was set at the MSY (Breuer, 2022), which means to catch the largest amount of fish over time while keeping the stock at the level of maximum production (Maunder, 2008). In addition, the use of multiannual management plans proposed in the previous reform of the CFP, became one of the most important tool for fisheries management in the European Union. Most of the existing multiannual management plans establish the management objective of exploiting species at their

MSY, as well as the corresponding measures to achieve them, which could include TACs and quotas, fishing effort restrictions and technical measures. The landing obligation was one of the most important changes introduced in the 2013 CFP reform (Salomon et al., 2014; Soto-Oñate and Lemos-Nobre, 2021). According to this regulation, all the stocks subject to a catch limit and/or a legal minimum conservation reference size must be landed, when caught, with the aim of reducing unwanted catches and eliminating discards (Prellezo and Villasante, 2023).

A.1.2 Mixed-fisheries management

In mixed-fisheries, several stocks are caught simultaneously (Wilson and Jacobsen, 2009). The catches of a mixed-fishery, are usually composed by stocks with different productivity and susceptibility to exploitation. Consequently, managing mixed-fisheries poses an additional challenge for managers, as the inability to distinguish between stocks makes it impossible to simultaneously achieve individual stock management objectives (García et al., 2019; Matsuda and Abrams, 2006; Ulrich et al., 2016). Either the advised catches for some stocks will be exceeded trying to catch the fishing opportunities of other stocks, or the fishing opportunities for some stocks will not be reached to prevent overshooting the fishing opportunities of the most restrictive stocks. Thus, one of the challenges of the CFP is the development of effective multiannual management plans for mixed-fisheries (Penas, 2016). Nowadays, the advice of fishing opportunities of stocks in mixed-fisheries multiannual management plans includes scenarios based on upper and lower values of fishing mortality around the fishing mortality at maximum sustainable yield (F_{MSY}). A range of fishing mortality around F_{MSY} objective is given to increase flexibility with the aim of achieving the MSY for all the stocks caught (ICES, 2022a). In Europe, four multiannual management plans for mixed-fisheries are currently in force. The first was adopted in 2016 for the fisheries and stocks in the Baltic Sea (EU, 2016). A multiannual management plan for the North Sea fisheries and stocks was adopted later, in 2018 (EU, 2018) and most recently, in 2019, two multiannual management plans came in force, one for stocks and fisheries in the Western Waters and adjacent waters (EU, 2019b) and another for those in the Western Mediterranean Sea (EU, 2019a).

Another challenge faced by the CFP in managing mixed-fisheries is the potential negative impact of the landing obligation on fleet activity. When the quota for a specific stock is reached and fishers are unable to avoid catching that stock while fishing for others, the fishing activity of the fleets must be stopped to prevent exceeding the stock's quota. As a results, this can lead to underutilization of the quotas allocated for the remaining stocks. The stocks that can potentially limit the fishing activity are the so-called choke stocks (Schrope, 2010). Improvements in selectivity have been postulated as possible solutions to overcome choking effects (Prellezo and Villasante, 2023). However, the selectivity changes of a mixed-fishery may only provide a partial solution to mitigate the potential impact of the landing obligation and the landing obligation may make the fishery activity unviable (Alzorriz et al., 2016). In anticipation of the potential negative consequences of the landing obligation in mixed-fisheries, the CFP proposed several flexibilities and exceptions (EU, 2013). These include the de minimis rule, which allows for discards of up to 5% of the total annual catches under certain circumstances, the species transfer rule which permits a deduction of up to 9% from the quota of the target species, and the year to year transfer rule which allows for the catching of up to 10% of the next year's quota in the current year. Additionally, there is an exception to the landing obligation that allows for the discarding of stocks with a high survival rate (EU, 2013). However, the problem of choke stocks in mixed-fisheries is not fully solved.

A.1.3 Decision making process for fisheries management in the European Union

Since the adoption of the Lisbon Treaty in 2007, the implementation of the CFP is the responsibility of the European Parliament and the European Council, comprised by ministers from the Member States (Casey et al., 2016). Although the setting of TACs is the sole responsibility of the European Council. The European Commission, initiates and proposes the legislation under the CFP, and also monitors its implementation (Figure A.4).

According to the article 6.2 of the CFP, Regulation (EU) No 1380/2013, the CFP requires “taking into account available scientific, technical and economic advice” (EU, 2013). Accordingly, the European Commission seeks the best available scientific advice for fisheries management from several scientific bodies (Figure A.4). The STECF (Scientific, Technical and Economic Committee for Fisheries) is the European Commission’s own scientific advisory committee and ICES (International Council for the Exploration of the Sea) is an independent intergovernmental marine science organization, that works on progressing and sharing scientific understanding of marine ecosystems and services to provide and to use this knowledge to ensure the sustainability of the seas and oceans. Recommendations and/or opinions from other stakeholders, such as those that come from fishing industry, non-governmental organizations (NGOs) or consumers, are also considered in the final proposal that the European Commission gives to the European Council and the European parliament (Figure A.4). These stakeholder organisations are grouped in Advisory Councils. Nowadays there are eleven Advisory Councils in Europe, depending on the region or the topic of interest (e.g., market, aquaculture, South Western Waters, pelagic, etc).

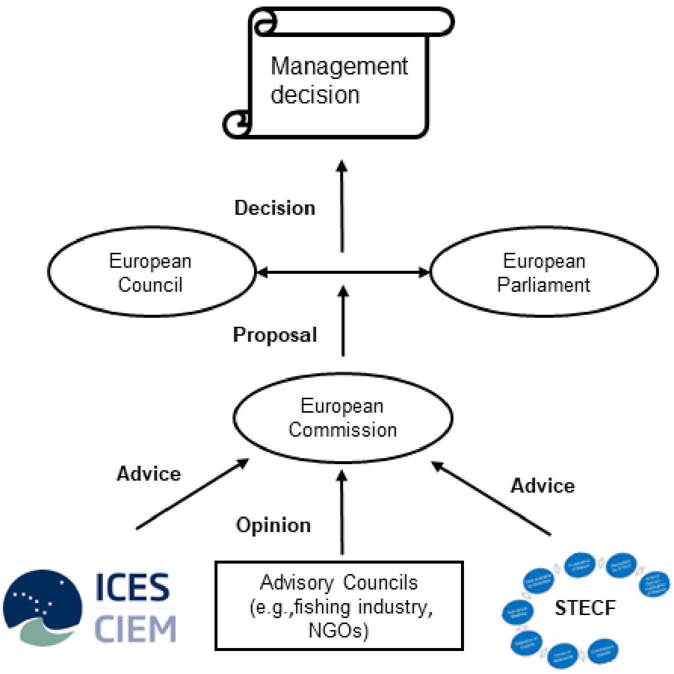


Figure A.4: The process of fisheries management decision making in the European Union.

A.2 ICES advice on fishing opportunities: from individual stock to mixed-fisheries considerations

The ICES advice on fishing opportunities is given at stock level to support stock-by-stock management objectives. The advice integrates the precautionary approach with the objective of achieving MSY (ICES, 2022a). In the absence of specific mixed-fisheries management objectives, ICES does not advise on specific mixed-fisheries catch opportunities for individual stocks, but provides mixed-fisheries considerations, where the impact of single-stock catch advice on the mixed-fisheries activity and on the sustainability of the stocks is evaluated under alternative fleet dynamics scenarios. This allows to assess the status of the stocks in the advised year under alternative fleet dynamics scenarios and to identify the most limiting stocks for the fleets.

A.2.1 Single stock advice

In 2012 ICES developed a framework to provide quantitative advice on fishing opportunities based on the data and knowledge available for each stock (ICES, 2012). In this framework the stocks are aggregated into six main categories according to available knowledge. ICES category 1 includes stocks with a quantitative stock assessment but the availability and quality of data decreases as one goes down in the categories (Table A.1). The stocks in ICES category 3 or greater are considered data-limited stocks (ICES, 2022a). In 2020, ICES provided single stock catch advice on more than 260 stocks on an annual or bi-annual basis and more than sixty percent of these stocks were data-limited stocks (ICES, 2020c).

Table A.1: Stock categories (Cat) on the basis of available knowledge (ICES, 2022a).

Cat	Description
1	Stocks with quantitative assessments; includes stocks with full analytical assessments and forecasts that are either age-/length-structured or based on production models.
2	Stocks with analytical assessments and forecasts that are only treated qualitatively as well as stocks with surplus production models; includes stocks with quantitative assessments and forecasts which, for a variety of reasons, are considered indicative of trends in fishing mortality, recruitment and biomass.
3	Stocks for which survey-based assessments or exploratory assessments indicate trends; includes stocks for which survey, trends-based assessment, or other indices and life-history information are available that provide reliable indications of trends in stock metrics such as total mortality, recruitment and biomass.
4	Nephrops stocks where information on possible abundance can be inferred and stocks for which a reliable time-series of catch can be used to approximate MSY. This is where there are reasonable scientific grounds to use life-history and density information from functional units to provide advice.
5	Stocks for which either only data on landings or a short time-series of catch are available.
6	Stocks for which there are negligible landings and stocks caught in minor amounts as bycatch; includes stocks where landings are negligible in comparison to discards as well as stocks that are primarily caught as bycatch species in other targeted fisheries.

ICES uses different methods and harvest control rules (HCRs) to provide management advice for each stock category. Overall, HCRs are mathematical formulas used to generate a recommendation about the catch limits of a fishery resource needed to achieve a pre-specified objective. For ICES category 1 and 2 stocks, ICES provides advice, when requested, in accordance with agreed management plans or strategies evaluated to be consistent with the precautionary approach (Figure A.5). If such plans are not agreed by the relevant management bodies or have been evaluated by ICES as not being precautionary, ICES gives advice for stocks in ICES category 1 and 2 using a HCR on the basis of the MSY approach (ICES, 2022a). The advice of stocks in category 3 is provided based on HCRs that use empirical indicators, such as abundance indices. These HCRs adjusts the advised catch with the trend observed on an available stock abundance index. In 2022, ICES began developing alternative low data demand HCRs to provide advice for stocks in ICES category 3 based on empirical indicators and MSY proxies (ICES, 2022a). For stocks in category 4 and greater, the advice of fishing opportunities is always given based on the precautionary approach (Figure A.5). This approach consists in giving an advice of fishing opportunities applying a precautionary buffer that reduces the catch advice periodically in a 20% without the use of any target reference points. As information on a stock status becomes more limited, more conservative advice should be given and a further margin of precaution should be adopted (ICES, 2022a).

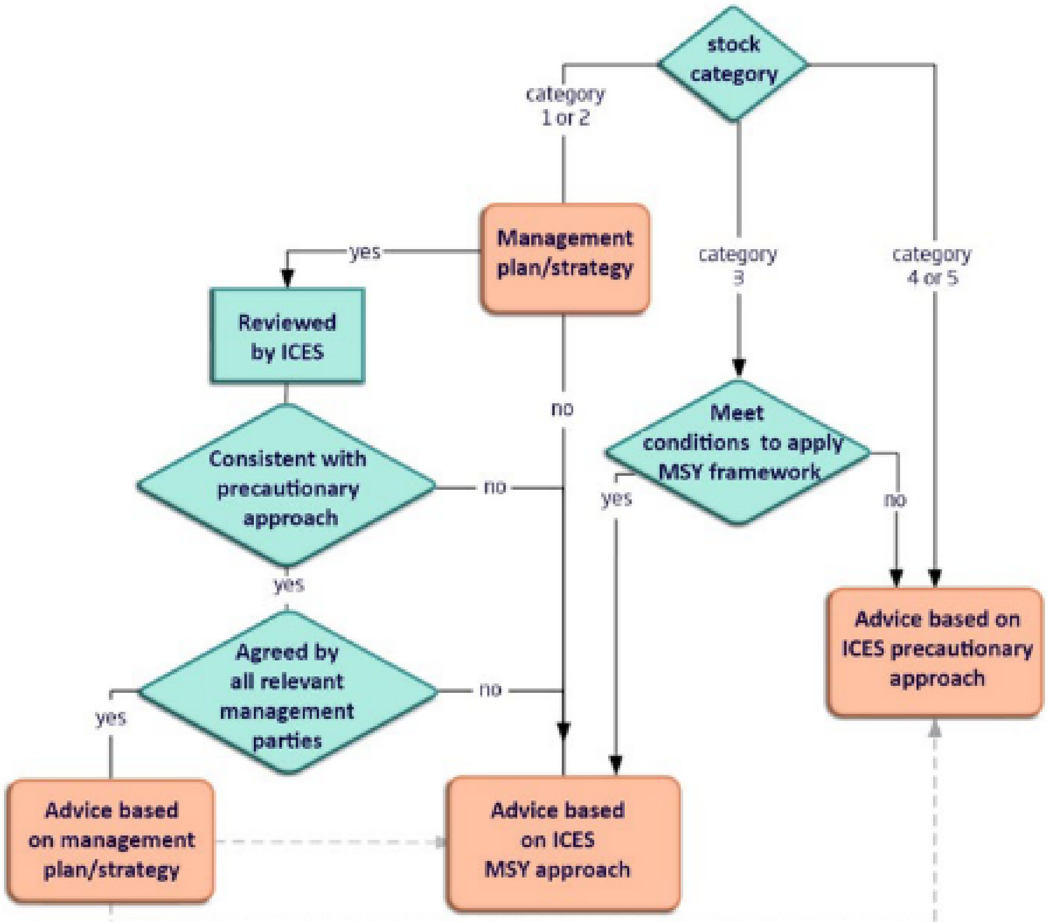


Figure A.5: Flow diagram showing the basis of ICES advice. The broken grey lines indicate that the advice in management plans is consistent with ICES MSY approach or the precautionary approach. Taken from ICES (2022a).

A.2.2 Mixed-fisheries considerations

The mixed-fisheries considerations provide an evaluation of the impact of single-stock catch advice on the mixed-fisheries activities and the overall sustainability of the stocks, considering various hypotheses regarding fleet dynamics. This consists in analysing alternative scenarios of fleet dynamics based on the current understanding of technical interactions among captured stocks and the catch advice for each stock. These considerations are not intended to generate a single, optimal catch advice for mixed-fisheries but rather to draw attention to the potential mismatches in single-stock catch advices in the context of mixed-fisheries. These mismatches occur because the catch advice of all the stocks cannot be complied at the same time in mixed-fisheries. In 2012, for the first-time in Europe, the mixed-fisheries considerations were provided by ICES for the North Sea demersal fishery (ICES, 2021j). Later, in 2015, 2016 and 2020 the mixed-fisheries considerations were developed for the Celtic Sea, Iberian waters and the Bay of Biscay respectively (ICES, 2021j). In 2022 mixed-fisheries considerations for the Irish Sea were presented for the first-time (ICES, 2022e). In addition, the development of mixed-fisheries considerations for the Baltic Sea are in process (ICES, 2022e,f).

A.3 Management strategy evaluation

The traditional approach for providing scientific advice for single stock fisheries management is based on a single hypothesis about the true system dynamics, which usually comes from the assessment model that best fits the data among those tested, and the knowledge about the population and the fishery (Punt and Donovan, 2007). At the mid-end of the 20$^{\text{th}}$ century, several international commercial fisheries collapsed, questioning the efficiency of the use of mathematical assessment and management models to base the advice on fishing opportunities (Schnute and Richards, 2001). Although there is not agreement about the concrete reason for these fisheries collapses in the scientific community, it is acknowledged that one of the main problems was the lack of the inclusion of uncertainty in the management process (Punt et al., 2016; Schnute and Richards, 2001).

As an alternative to the traditional approach, the Management Strategy Evaluation (MSE) was first used at the end of the 20$^{\text{th}}$ century to provide a framework for the inclusion of uncertainty in the decision making process. The MSE, consists in evaluating the performance of alternative management strategies on stocks and fisheries sustainability under different hypotheses about the system dynamics including different sources of uncertainties. The aim of the MSE is to identify a management strategy that meets pre-specified objectives while being robust to the system's uncertainties (Punt et al., 2016). The introduction of the MSE supposed a change of paradigm. Instead of looking for the model that best fitted to the data available, the objective became to find a management procedure that is robust to the uncertainty in the fishery system.

The MSE, pioneered by the International Whaling Commission (IWC) in 1980, was introduced into fisheries lexicon by Smith in 1994 (Smith, 1994). However, the use of MSE in fishery research was popularized in the 21$^{\text{st}}$ century, with the revolution in computational power. Nowadays the MSE is used worldwide considering it as the most appropriate way to test a multiannual management plan (Punt et al., 2016).

In addition, MSE improves the transparency of the scientific advice and the communication between scientists and stakeholders (deReynier et al., 2010) reducing the influence of politics in the decision making process (Punt and Donovan, 2007).

Among the available software for MSE, FLBEIA (Bio-Economic Impact Assessment using FLR libraries) (García et al., 2017) is one of the mixed-fisheries bio-economic simulation models available to support management decisions in Europe (ICES, 2022f). The FLBEIA model is developed in R (R Core Team, 2022) using FLR libraries (Kell et al., 2007). FLBEIA model is formed by two main components which interact among themselves (Figure A.6), namely the Operating Model (OM) and the Management Procedure Model (MPM). The OM simulates the real system, and it is composed by stocks, fleets, and their interaction. The MPM is the part of the model that simulates the management process, and it is composed by data, perceived system and advice, generated by the observation model, the assessment model and the HCR, respectively. The link between OM and MPM is done through the observation model, that generates the observed data from the OM to be used in the assessment model of the MPM. The link between MPM and OM is done through the implementation model, which describes how the management advice of the MPM is implemented in reality. The model is constructed in such a way that allows to account for the major sources of uncertainties of the fisheries management system as described by Francis and Shotton (2011).

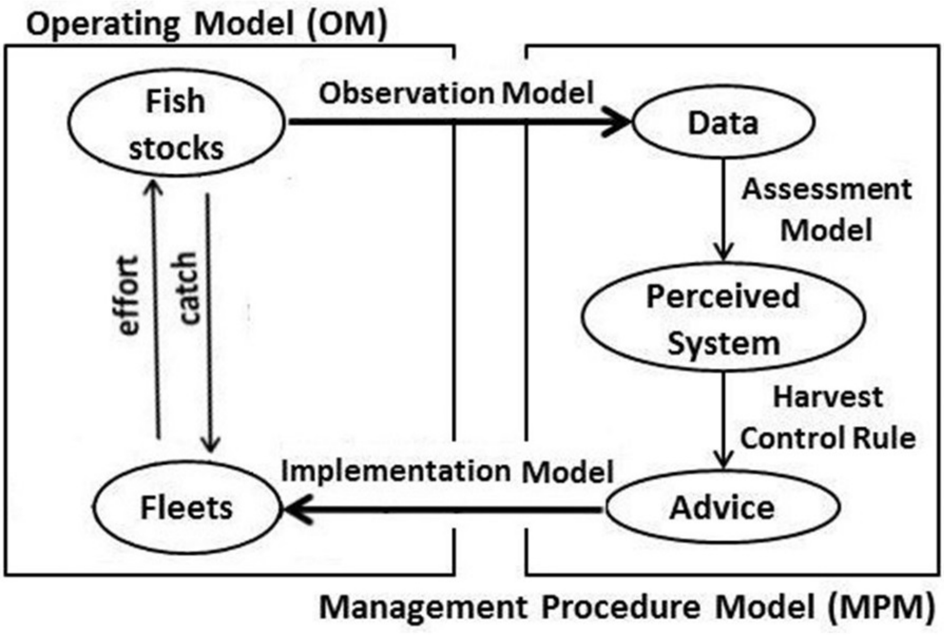


Figure A.6: Conceptual diagram of the management strategy evaluation framework. Adapted from Garcia et al. (2013).

Lately, there is controversy about what simulation study can be defined as an MSE implementation and what not. Some authors consider that management strategy simulation models, where the catch advice derived from the HCR is perfectly implemented in reality, cannot be considered MSE (Punt et al., 2016). In single stock approaches, the difference usually arises from the introduction of an assessment model in the management procedure to obtain the perceived system.

The first applications of MSE were single-stock (e.g., Garcia et al., 2011; Polacheck et al., 1999). However, in the last decades, the need of a more holistic approach to fisheries management in the light of Ecosystem-Based Fisheries Management (EBFM), led to multi-stock and multi-fleet approaches to be implemented (e.g., García et al., 2019; Simons et al., 2014). In addition, there have been attempts to use full end-to-end ecosystem model as OM in MSE approaches (e.g., Dichmont et al., 2016b; Fulton et al., 2016) in an attempt to transition towards EBFM (Lidström and Johnson, 2020).

A.3.1 Challenges for mixed-fisheries MSE

Mixed-fisheries catch a large number of species, making it practically impossible to include all of them in the simulation model explicitly. This is primarily due to the significant workload, computational resources needed, and limitations in available data. However, the number of species included in the simulation should be large enough to be able to describe the activity of the fleets realistically. This ensures that the impact of a management strategy on biological sustainability can be accurately assessed (Ulrich et al., 2011). The stocks included in the operating model will determine how well the fishing activity can be described and what we can be assessed in terms of the biological sustainability. Consequently, one of the challenges to build a multi-stock MSE is to identify the stocks to be included in the simulation. Once the selection of the stocks is done, the next step is to condition their population dynamics to be able to incorporate them in the MSE simulation. Of the stocks caught in a mixed-fishery most of them are bycatch species for which data available is very limited. Many multi-stock MSE approaches have been focused on data-rich stocks and the few data-limited stocks included were treated over-simplistically (e.g., Prellezo et al., 2016; STECF, 2015). Determining the most plausible dynamics of data-limited stocks is one of the major challenges to build a MSE.

A.3.1.1 The selection of stocks

The selection of the stocks to be included in the simulation should be done in the context of the three pillars of the sustainability –the social equity, the economic viability and the ecological protection (WCED, 1987) –. However, in many cases, the stocks included in the simulation model are selected based exclusively on one of these considerations. There are MSE studies focused on the economically most important stocks for the fishery (e.g., Hamon et al., 2007; Prellezo et al., 2016) and studies that include in the MSE those stocks that need to be ecologically protected (e.g., Bastardie et al., 2010; Punt et al., 2001).

A framework to assess the ecological risk for the effects of fishing was developed by Hobday et al. (2011, 2007). The framework divides ecological risk assessment models in three groups depending on their data needs. Level 3 comprises fully quantitative approaches, level 2 includes semi-quantitative approaches and level 1 consists of largely qualitative approaches (Hobday et al., 2011, 2007). The Productivity-Susceptibility Analysis (PSA) is a semi-quantitative low data demand risk assessment model that falls within the level 2 approaches. The potential risk is derived from two characteristics: the productivity, which determines the capacity of a stock to recover from depletion, and the susceptibility, which quantifies the potential impact of the fishing activities on the stock. The basis of the PSA is that lower productivity and higher susceptibility to the fishery imply higher potential risk for the stock (Hobday et al., 2007). The PSA has demonstrated its value to provide relative potential risk of all the stocks within the same framework despite of their differences in data availability (Cortes et al., 2015; Hobday et al., 2011; Lucena-Fredou et al., 2017; Ormseth and Spencer, 2011).

Osio et al. (2015) proposed a risk based approach to prioritise data collection and research efforts in Mediterranean Sea demersal fisheries. They developed a ranking system for stocks based on their contribution to landings, their average price per kilogram, and their ecological risk for the effects of fishing, which was calculated using PSA.

A.3.1.2 Conditioning of data-limited stocks

In multi-stock MSE implementations the emblematic stocks, including data-limited stocks, should be included explicitly. Some of the studies that include the population dynamics of data-limited stocks in a simulation model use life-history parameters, an assumption about population biomass in the first historical year, and an assumption about the historical exploitation to parameterise the population dynamics equations (e.g., Carruthers et al., 2014; Fischer et al., 2020). An alternative is to use a low data demand assessment model to include the population dynamics of data-limited stocks in a simulation model. For example, biomass-dynamics models (e.g., Harlyan et al., 2019; Mildenberger et al., 2022), or age-structured models (e.g., Geromont and Butterworth, 2014; Huynh et al., 2020b).

In recent years the popularity of biomass-dynamics models have increased in ICES community to assess the status of data-limited stocks (ICES, 2020c). Surplus Production model In Continuous Time (SPICT, by Pedersen and Berg, 2017) and Bayesian State–Space biomass-dynamics model (JABBA, by Winker et al., 2018) are the models most used by ICES to assess the population dynamics of data-limited stocks (ICES, 2020c). Biomass-dynamics models represent the dynamics of fish stocks simplistically. They ignore the age-structure and try to explain changes in abundance as a function of the removal of biomass by fishing, the biomass in the previous year, and the growth in biomass. Furthermore, they make strong assumptions about the nature of the system being modelled and the data used to fit the models (see Punt and Hilborn, 1996, to view such assumptions). The results are sensitive to the assumed production function (Maunder, 2003) and to the impacts of transient age-structured (Punt and Szuwalski, 2012). Overall, age-structured stock assessment models are preferable alternatives to biomass-dynamics models because they allow the inclusion of more detail in the population dynamics such as, age-structured selectivity, recruitment, and individual growth (Maunder, 2003). The majority of contemporary stock assessment models are based on age-structured models (Punt et al., 2013). However, such models cannot easily be applied to data-limited stocks because most of them need catch-at-age data and these data are unavailable for most of the data-limited stocks.

The Stock Reduction Analysis (SRA) model was developed for situations where catch-at-age data are not available (Huynh et al., 2020a; Kimura and Tagart, 1982; Walters et al., 2006). The SRA model is a low data demand age-structured assessment model that combines the available catch and abundance index data with life-history parameters to derive historical estimates of population biomass and exploitation level. It consists of finding the combination of initial biomass, fishing mortality and recruitment levels that could have generated the observed data given a specific set of values for biological parameters, standard deviation of the recruitment deviates and survey and fishery selectivities (Huynh et al., 2020a). The minimum data required to apply the SRA model is annual total catch and an abundance index of total biomass. Additionally, SRA model can use other data sources such as annual mean length in the catch, catch-at-length, catch-at-age and index-at-age data. Moreover, it allows to include uncertainty in all the life-history parameters and data sources. The SRA model, has been used in several studies to define the population dynamics of data-limited stocks in a simulation model (Hordyk et al., 2019; Huynh et al., 2020b). Hordyk et al. (2019) evaluated the performance of the SRA model under bias in biological parameters and error in observation and model assumptions. However, the SRA model's ability to assess the population dynamics under different data availability scenarios have never been quantitatively evaluated.

A.4 Rationale for the study

The Bay of Biscay is a well-differentiated geomorphological unit orientated toward the North-West and located in the temperate Northeast Atlantic ocean, south of Celtic Sea (Lavín et al., 2006) (Figure A.7). The Bay of Biscay is limited by Spanish coast in the south and French coast in the east. It forms the transitional region between the cold boreal waters and the warm waters of the temperate biogeographical province (Lavín et al., 2006), which results in a higher biodiversity in comparison to adjacent areas making it perfect for multispecies fisheries (Prellezo et al., 2016).

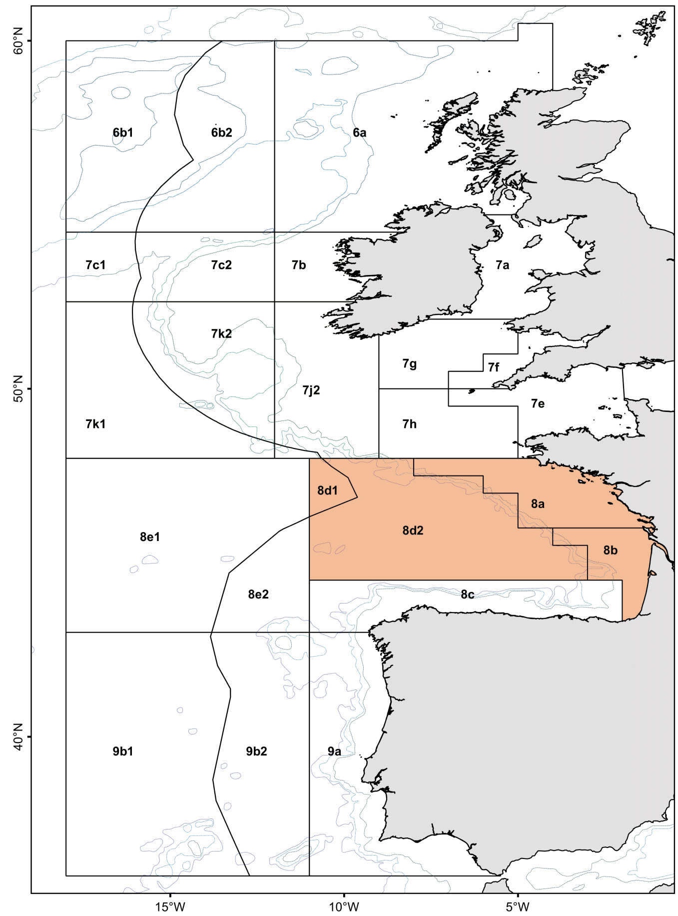


Figure A.7: Western Europe marine area divided by ICES divisions. Shaded orange area is the study area in the Bay of Biscay.

The management of the stocks caught in the Bay of Biscay is conducted under the CFP and regional legislations. The main fisheries management tools in this area are TACs and quotas (ICES, 2021a). However, there are also technical measures outlined in Regulation (EU) No 2019/1241, which aim to ensure the conservation of fishery resources and the protection of marine ecosystems. These measures apply to certain stocks and fishing gears. Among technical measures, there are minimum landing size, mesh size limitation and marine protected areas. The minimum conservation reference size is implemented for 41 species in the Bay of Biscay. The mesh size is limited for trawlers, static nets and driftnets at different size based on the target stocks of the gear. The landing obligation was introduced for pelagic stocks in 2015, for demersal stocks in 2016 and has been fully in force since January 2019 (ICES, 2021a). As an exception, rays are exempted to the landing obligation because they have a high survival rate after discarding, and hake, megrim, anglerfishes, sole, mackerel and horse mackerel are subject to de minimis exemptions which allow discards up to a 5% of the total annual catches of these stocks (EU, 2023). Some of the demersal stocks in the Bay of Biscay area are included in the Western Waters and adjacent waters multiannual management plan (EU, 2019b). The objective of the plan is to reduce unwanted catches, to minimize negative fishing impacts on the marine ecosystem and to contribute to the achievement of the CFP objectives by applying the precautionary approach to fisheries management and by ensuring that harvested populations are above levels which can produce MSY.

In the Bay of Biscay there is a long tradition of demersal mixed-fisheries (Prellezo, 2010; Prellezo et al., 2016). These fisheries are predominantly conducted by French and Spanish vessels, which collectively account for over 86% of the total catches in the area. Specifically, approximately 53% of the catches are attributed to around 1,500 French vessels, while 33% can be attributed to 57 Spanish vessels that operate within the area. The rest of the catches correspond to vessels from Belgium, Germany, Denmark, Ireland, Netherlands, Poland, Portugal and United Kingdom. The French and Spanish fisheries vessels are mostly bottom and mid trawlers, gillnets and longliners. The main species caught by these vessels are hake, anglerfish, megrim, sole, Norway lobster, sea bass and pollack (ICES, 2021a). Other target species are horse mackerel, mackerel, blue whiting, sea bass, cuttlefish and squids. The composition of species caught in demersal mixed-fisheries in the Bay of Biscay varies depending on the fishing area, depth range, and gear employed. Overall, the demersal mixed-fisheries operating in the Bay of Biscay catch more than 150 species (Altuna-Etxabe et al., 2020; Briton et al., 2020). However, only approximately one-fourth of these species are subject to TAC and quota regulation (Briton et al., 2020). Among the species included in the TAC and quota system for these fisheries in the Bay of Biscay, there are hake, megrim, white anglerfish, black anglerfish, whiting, Norway lobster, sole, mackerel, horse mackerel, blue whiting, pollack, cuckoo ray, thornback ray and undulate ray (EU, 2022) (Table A.2). The first eleven species are included in the ICES Bay of Biscay mixed-fisheries considerations (ICES, 2022b), while the first seven species together with sea bass are included in the Western Waters and adjacent waters multiannual management plan (EU, 2019b). Most of the other species caught are data-limited with unknown status and without any quota restriction (Table A.2).

In 2015 the STECF evaluated the potential impact of the Western Waters and adjacent waters multiannual management plan on the demersal mixed-fisheries operating in the Bay of Biscay and on their fishery resources (STECF, 2015). Two simulation

Table A.2: Information of the stocks caught by the demersal mixed-fisheries in the Bay of Biscay that are included in the TAC and quota system and/or in the Western Waters and adjacent waters multiannual management plan. Stock common and scientific name, distribution area using ICES nomenclature, ICES data category (Cat), advice type (MSY approach or precautionary approach (PA)), status with regards to MSY reference points, and information if the stocks are included in the TAC and quota system (TAC) and/or in the Western Waters and adjacent waters multiannual management plan (MP) and/or in the Bay of Biscay mixed-fisheries considerations (MF) and the reference. In the status column, grey represents unknown reference points, green represents either a stock that is fished below F_{MSY} and/or whose size is above MSY $B_{trigger}$; red represents either a stock that is fished above F_{MSY} and/or whose size is below MSY $B_{trigger}$.

Common name	Scientific name	Stock distribution area	Cat	Advice	Status		Management			Reference
					F	B	TAC	MP	MF	
Hake	*Merluccius merluccius*	Subareas 4, 6 and 7, and divisions 3a and 8abd	1	MSY	✓	✓	X	X	X	ICES (2021c)
Megrim	*Lepidorhombus whiffiagonis*	Divisions 7b-k and 8abd	1	MSY	✓	✓	X	X	X	ICES (2021g)
White anglerfish	*Lophius piscatorius*	Subarea 7, and divisions 8abd	1	MSY	✓	✓	X	X	X	ICES (2021o)
Black anglerfish	*Lophius budegassa*	Subarea 7, and divisions 8abd	1	MSY	✓	✓	X	X	X	ICES (2022c)
Whiting	*Merlangius merlangus*	Subarea 8, and division 9a	5	PA	?	?	X	X	X	ICES (2021p)
Norway lobster	*Nephrops norvegicus*	Divisions 8ab	1	MSY	✓	?	X	X	X	ICES (2021h)
Sole	*Solea solea*	Divisions 8ab	1	MSY	✗	✗	X	X	X	ICES (2021m)
Mackerel	*Scomber scombrus*	Subareas 1-7 and 14, and divisions 8a-e and 9a	1	MSY	✓	✓	X		X	ICES (2021f)
Horse mackerel	*Trachurus trachurus*	Subarea 8, and divisions 2a, 4a, 5b, 6a and 7a-c,e-k	1	MSY	✓	✗	X		X	ICES (2021d)
Blue whiting	*Micromesistius poutassou*	Subareas 1-9, 12 and 14	1	MSY	✗	✓	X		X	ICES (2021b)
Pollack	*Pollachius pollachius*	Subarea 8, and division 9a	5	PA	?	?	X		X	ICES (2021i)
Cuckoo ray	*Leucoraja naevus*	Subareas 6-7, and divisions 8abd	3	PA	?	?	X			ICES (2020a)
Thornback ray	*Raja clavata*	Subarea 8	3	PA	?	?	X			ICES (2020d)
Undulate ray	*Raja undulata*	Divisions 8ab	6	PA	?	?	X			ICES (2020e)
Sea bass	*Dicentrarchus labrax*	Divisions 8ab	1	MSY	✓	✓		X		ICES (2021k)

models were implemented: one focused on the French fleets and another on the Spanish fleets. In the French simulation model, the operating model included 17 species that covered 56% of the total landings of the French fleets and more than 68% of the total revenue of the French fleets, whereas in the Spanish simulation model the operating model included 12 species covering the 81% of the total catches of the Spanish fleets and more than 88% of the total revenue of the Spanish fleets (STECF, 2015). From these stocks, only three species (hake, sole and nephrops) in the French model and two species (hake and megrim) in the Spanish model were simulated using an age-structured population dynamics (STECF, 2015). The rest of the species were simulated using a constant Catch Per Unit Effort (CPUE) approach, whereby the catch is linearly related to effort and is independent of the biomass (STECF, 2015). The role of the species included with the constant CPUE approach was to be able to restrict the fishing activity while contributing to the yield of the fleets. However, without including their population dynamics in the simulation model, it is not possible to assess the impact of a management strategy plan on their sustainability.

This thesis is focused on the improvement of the simulation model implemented in 2015 by STECF (STECF, 2015) for the evaluation of the multiannual management plan of the demersal mixed-fisheries in the Bay of Biscay. On the one hand, the dynamics of French and Spanish vessels in the operating model will be included, based on the fleets and metiers segmentation used by ICES for the provision of Bay of Biscay mixed-fisheries considerations (ICES, 2022b,e). On the other hand, the population dynamics of more stocks, both data-rich and data-limited, will be included in the simulation algorithm. This will require giving special attention to the selection of the stocks to be included in the simulation and to the conditioning of data-limited stocks. The selection of stocks will be based on a proposed extension of the priority ranking developed by Osio et al. (2015), while the conditioning of data-limited stocks will be based on the SRA model (Huynh et al., 2020a). The improved simulation algorithm will be implemented in the FLBEIA model (García et al., 2017) and will be used to assess the biological sustainability of current and alternative management strategies for demersal mixed-fisheries in the Bay of Biscay considering the implementation or non-implementation of the landing obligation. Overall, this work will contribute to provide a holistic evaluation of the management of the mixed-fisheries operating in the Bay of Biscay towards one of the main objectives of the CFP which is to ensure the sustainability of stocks caught while maintaining the fishing activity in the long-term.

A.5 Working hypothesis and objectives

The working hypothesis (Dewey, 1938; Shields and Tajalli, 2006) that guides the research conducted during this PhD is:

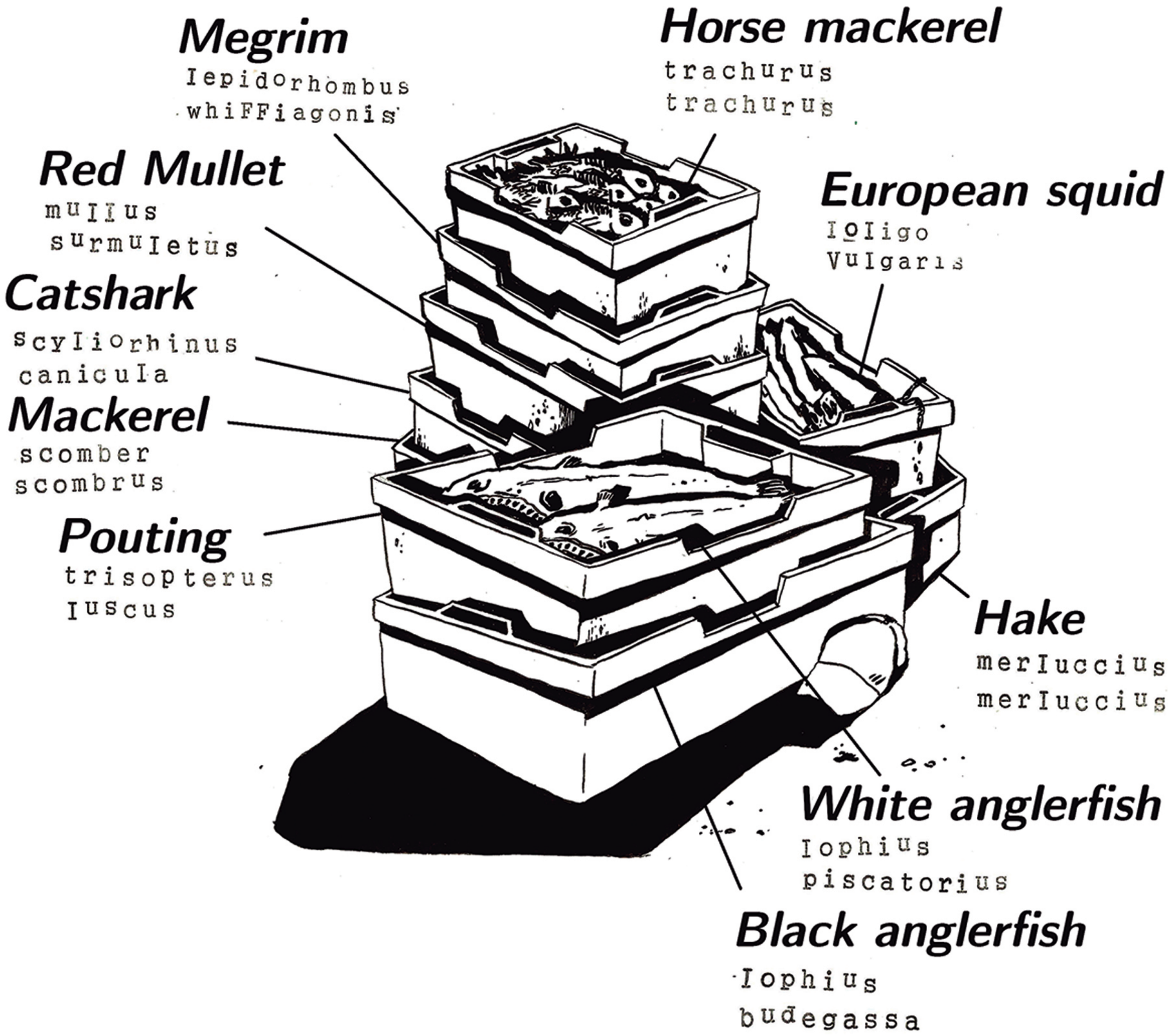
"The evaluation of the multiannual management plan for the demersal mixed-fisheries operating in the Bay of Biscay requires an adequate stock selection and a reliable representation of the data-limited stock dynamics to provide a holistic assessment of the performance of the management strategies regarding the sustainability of the most emblematic stocks".

The veracity of the working hypothesis has been evaluated through three specific research objectives:

1. To identify the most at risk, most exploited and/or economically most important stocks for the fishery that should be included in the management strategy evaluation framework.
2. To evaluate by "self-test" simulation the ability of the SRA model to obtain accurate population dynamics estimates under alternative data availability scenarios.
3. To evaluate the impact of current and alternative management strategies for the demersal mixed-fisheries operating in the Bay of Biscay on the sustainability of the most emblematic stocks.

A.6 Structure of the thesis

The thesis starts with a general introduction provided in this chapter, Chapter 1, about the main challenges in the management of mixed-fisheries in the European Union, and in the conditioning of multi-stock simulation models. Following this, there are three chapters, each with its own introduction, materials and methods, results and discussion sections. Each of these chapters addresses a specific research objective. First, those stocks that should be included in the simulation algorithm are identified in Chapter 2. The selection of the stocks is based on an extension of the species prioritisation approach developed by Osio et al. (2015). The stocks are selected based on the risk for the effects of fishing activity calculated using a PSA approach (Hobday et al., 2011), the catch level and the economic importance. In Chapter 3, the potential use of the SRA model (Huynh et al., 2020a) to obtain accurate population dynamics estimates under alternative data availability scenarios is tested by simulation. This SRA model is the model used to condition the population dynamics of data-limited stocks in the simulation model in Chapter 4. The simulation using FLBEIA model (García et al., 2017), including the conditioning of all selected data-rich stocks based on their assessment model estimates and some selected data-limited stocks based on the SRA, is described in detail in Chapter 4. The simulation provides an assessment of the performance of the current and alternative management strategies in terms of stocks sustainability considering the implementation or non-implementation of the landing obligation. The main limitations and major findings of all the previous chapters, together with the future potential work, are discussed in Chapter 5. The main conclusions are summarized in Chapter 6. A list of the scientific contributions carried out during this thesis is given in Chapter 7. Finally, supplementary material for the three main chapters is provided in Appendix C, Appendix E and Appendix F.



Appendix B

Identification of the species that should be included in the MSE

B.1 Introduction

Before a multiannual management plan is put in place, an impact assessment of the management measures on the system should be carried out to ensure that it produces the desired outcomes. However, carrying out an impact assessment of mixed-fisheries management plans is a great challenge because the amount of species caught by these fisheries is high and the number of species included in the impact assessment should be large enough to describe the activity of the fishery realistically. Thus, the first step to implement the simulation model is to identify the species that should be included in order to ensure that the achievement of the management objectives can be assessed.

The need to assess the ecological sustainability of fisheries worldwide has led to the development of a risk assessment models (Hobday et al., 2007; MSC, 2001). The PSA is one of them (Hobday et al., 2011, 2007). The PSA is a semi-quantitative low data demand risk assessment model. It was first developed by Stobutzki et al. (2001) to assess the risk of bycatch species due to trawling in the Australian Northern prawn fishery. The risk level identified by the PSA is used to prioritise the species at highest risk for a fully quantitative assessment (Hobday et al., 2007). Due to its flexibility and minimal data requirements, it has been applied in a wide range of fisheries worldwide from target to bycatch species; the latter covering different groups such as seabirds, sea turtles, sharks, rays, and marine mammals (Cortes et al., 2015; Lucena-Fredou et al., 2017; Okemwa et al., 2016; Waugh et al., 2012). Furthermore, it has been applied not only to species but also to different ecosystem components such as communities and habitats (Williams et al., 2011).

The objectives of this chapter are to assess the potential risk of the species caught by the Basque demersal mixed-fisheries, and to identify the most at risk, most exploited, and/or economically most important species for the Basque demersal mixed-fisheries. First, we conducted a PSA of the species exploited by this fishery (Hobday et al., 2011). Then, we ranked the species according to their potential risk (PSA), their revenue, and their catch volume modifying the prioritisation approach proposed by Osio et al. (2015). Finally, we discussed the potential use of this rank to decide on the species that should be included in the evaluation of the Basque demersal mixed-fisheries management plan.

B.2 Materials and methods

B.2.1 Species selection

The Basque demersal fishery is a mixed-fishery that operates in the western EU waters, from the Cantabrian coast to the west of Scotland. However, this study is focused solely on the Bay of Biscay (ICES divisions 8abd) (Figure A.7), where most of the reported landings of the fishery occur (approximately 80% of total landings between 2001 and 2017). It is composed of three fleets that operate with different gears: bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). OTB fleet operates at depths of 10–1,400 m with a mesh size of 70 mm, PTB fleet operates at depths of 24–400 m with a mesh size of 100 mm, and LLS fleet operates below 300 m deep with 1,000 hooks per longliner.

The activity of the fishery is divided in six metiers depending on the gear and target species (Iriondo et al., 2010) (Table B.1). We analysed the species composition of their landings from 2001 to 2017 using port sampling and their discards based on observer information from 2003 to 2017. Overall, the landings and discards of the six metiers included approximately 150 species. We selected the species that contributed up to the 95% of the total landings plus the total sampled discards by metier. In addition, to ensure the inclusion of the species that could constrain the activity of the Basque demersal mixed-fisheries under the landing obligation, we included all the species subject to catch restriction even if their landing and discard contributions were not within the 95%. Fleet landings and discards information was obtained from the AZTI fisheries database, that includes port sampling and observer data.

Table B.1: Description of the six metiers that compose the Basque demersal mixed-fisheries according to fishing gear and target species. Gears: bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS).

Métier	**Gear**	**Target**	**Species**
OTB_DEF	OTB	Demersal fish	Hake, megrim, four-spot megrim, white anglerfish, black anglerfish, pouting
OTB_MCF		Mix of cephalopod and demersal fish	European squid, veined squid, cuttlefish, elegant cuttlefish, pink cuttlefish, pouting, red mullet
OTB_MPD		Mix of pelagic and demersal fish	Hake, mackerel, horse mackerel
OTB_SPF		Small pelagic fish	Mackerel, horse mackerel
PTB_DEF	PTB	Demersal fish	Hake
LLS_DEF	LLS	Demersal fish	European conger, hake

B.2.2 Productivity-susceptibility analysis

We carried out a PSA for the species selected. Productivity is determined by demographic characteristics and was calculated for each species (Hobday et al., 2011). Susceptibility is determined by the interaction of fisheries and species; as the Basque demersal mixed-fisheries are composed of different gears with different susceptibilities, first, we assessed the susceptibility of each species and gear (OTB, PTB and LLS) independently (Hobday et al., 2011); then, we calculated the cumulative impact of overlapping fishing activities using the aggregated susceptibility method proposed by Micheli et al. (2014). The productivity and susceptibility characteristics (called attributes) were scored between 1 (low risk) and 3 (high risk), and potential risk scores were calculated (Hobday et al., 2007). Those scores were displayed graphically on an x–y plot (i.e., the PSA plot).

B.2.2.1 Productivity

The productivity attributes selection is based on the PSA proposed by Hobday et al. (2011), which scores seven productivity attributes: maximum length, maximum age, length at maturity, age at maturity, fecundity, reproductive strategy, and trophic level. Of those attributes, we excluded the trophic level attribute (Duffy and Griffiths, 2017; Hordyk and Carruthers, 2018). Although some authors (Hobday et al., 2007; Patrick et al., 2010) relate trophic level inversely with productivity, in our case study trophic level could not be representative of the productivity of some of the species analysed. For example, cephalopods, species from Sepiidae, Octopodidae and Ommastrephidae families, show high trophic level due to cannibalism but are also very productive.

The productivity attributes were scored between 1 (low risk or high productivity) and 3 (high risk or low productivity). Reproductive strategy thresholds were maintained as in Hobday et al. (2011); live bearers (low productivity), demersal egg layer (medium productivity), and broadcast spawners (high productivity) assuming that the species that are broadcast spawners have the capacity to produce more young so they have higher capacity to recover if their population size is reduced compared with species that bear live (Stobutzki et al., 2001). The basis of this categorization is that fast-growing species have higher capacity to recover if their population size is reduced. High productivity has small, short lifespan, early mature and larger fecundity species (Winemiller and Rose, 1992). For the other attributes we tested two alternative methods to define the thresholds: the quantile method that divides the species in equally size groups (Lucena-Fredou et al., 2017) and the K-mean clustering method that group the most similar species respect to the variance of each species attribute value and each group mean value (Patrick et al., 2009). The scoring thresholds of each attribute using each method are presented in Table B.2.

Table B.2: Productivity attributes thresholds adapted from Hobday et al. (2011). The attribute values were divided into low, medium, and high productivity categories. Reproductive strategy: fish which release their gametes into the water (Broadcast spawners – External fertilization, BS), species that lay eggs (oviparity, egg layers – Internal fertilization, DS), and ovoviviparity and viviparity species (Live bearers – Internal fertilization, LB)

Productivity attributes	**Low productivity (High risk, score =3)**		**Medium productivity (Medium risk, score =2)**		**High productivity (Low risk, score =1)**	
	K-mean	Quantile	K-mean	Quantile	K-mean	Quantile
Age at maturity (year)	>6.66	>5	2.78–6.66	2.5–5	<2.78	<2.5
Maximum age (year)	>31.8	>20	11.5–31.8	12–20	<11.5	<12
Length at maturity (cm)	>124	>35.4	50.5–124	20–35.4	<50.5	<20
Maximum length (cm)	>163	>90	69.3–163	46–90	<69.3	<46
Fecundity (min egg/year)	<3.7E+5	<4.5E+3	3.7E+5–1.8E+6	4.5E+3–8.8E+4	>1.8E+6	>8.8E+4
Reproductive strategy	LB		DS		BS	

All the productivity attributes were weighted equally, and the overall productivity score was calculated as the average of all the productivity attributes scores (Hobday et al., 2011).

B.2.2.2 Susceptibility

Individual gears Susceptibility was estimated based on the four susceptibility attributes proposed by Hobday et al. (2011): availability or horizontal overlap between fleet distribution and species distribution, encounterability or vertical overlap between fleet distribution and species distribution, selectivity or length distribution caught by the fishery when encountered, and post-capture mortality or species capacity to survive after release. Those level 1 attributes were defined by the combination of level

2 attributes. Availability is defined according to the maximum score of three level 2 attributes; global distribution, migration capacity and aggregation behaviour of the species, and encounterability is defined according to the minimum score of two level 2 attributes; accessibility of the gear to the species adult habitat and bathymetry overlap between fishery and species distribution.

Each of the level 2 attribute was scored between 1 (low risk or low susceptibility) and 3 (high risk or high susceptibility). Those scoring criteria were modified to better suit the Basque demersal mixed-fisheries and species characteristics (Table B.3). The thresholds of adult habitat overlap, and bathymetry overlap attributes were defined independently per each gear (Table C.1 in Appendix C). Selectivity thresholds were based in Marine Stewardship Council (Hordyk and Carruthers, 2018; MSC, 2001), that defines selectivity attribute according to 3 risk categories: low risk when length at capture (Lc) is greater than length of maturity (Lm), medium risk when 0.5Lm <Lc <Lm, and high risk when Lc <0.5Lm. Post-capture mortality was defined in a general way, assuming that sharks and rays have higher survival capacity comparing with other species.

Table B.3: Susceptibility attribute thresholds adapted from Hobday et al. (2011). The attribute values were divided into low, medium, and high susceptibility categories. The definitive availability score was the maximum score of the global distribution, migration and aggregation attributes. For encounterability, the minimum score of adult habitat and bathymetry was used. Encounterability and selectivity attributes were calculated per each gear (Table C.1 in Appendix C). Lc, length at 50% selectivity. Lm, length at 50% maturity.

Susceptibility attributes		**Low susceptibility (Low risk, score =1)**	**Medium susceptibility (Medium risk, score =2)**	**High susceptibility (High risk, score =3)**
Availability	Global distribution	Worldwide (W)	North Atlantic (NA)	Bay of Biscay (BoB)
	Migration	Highly migratory	Few restrictions to dispersal	Species can not complete its life-history at sea
	Aggregation	No seasonal peaks in feeding, mating, spawning	Some seasonal peaks but breeding not restricted to a particular season	Species forms breeding aggregations
Encounterability	Adult habitat	Low overlap with fishing gear	Medium overlap with fishing gear	High overlap with fishing gear
	Bathymetry	Low overlap with fishing gear	Medium overlap with fishing gear	High overlap with fishing gear
Selectivity		Lc >Lm	0.5Lm <Lc <Lm	Lc <0.5Lm
Post-capture mortality		Released alive	Evidence of release some alive	Released die

All susceptibility level 1 attributes were weighted equally, and the overall susceptibility score was calculated as geometric mean (Hobday et al., 2011). In the cases where a fleet had no catches associated to a species (Table C.3, Table C.4, and Table C.5 in Appendix C), we assumed there were external factors that made the likelihood of catch equal to 0, avoiding the catch of this species. Thus, those species are not displayed graphically on the corresponding PSA plot.

Overall fishery The cumulative impact of overlapping fishing activity was calculated using the aggregated susceptibility (AS) proposed by Micheli et al. (2014):

$$AS = min\left(3, 1 + \sqrt{(S_{OTB} - 1)^2 + (S_{PTB} - 1)^2 + (S_{LLS} - 1)^2}\right), \tag{B.1}$$

where S_{OTB}, S_{PTB}, and S_{LLS} are the susceptibility scores for the OTB, PTB, and LLS fleets, respectively. Therefore, the AS index increases with the number of fisheries, and the cumulative potential impact may be larger than that generated by a single fleet (Halpern et al., 2008). The minimum and maximum values are equal to 1 and 3, respectively, to maintain consistency with the range of susceptibility scores of the original PSA index (Micheli et al., 2014).

B.2.2.3 Potential risk

The potential risk (R) for each species was calculated, for each single gear and for the overall fishery, as the Euclidean distance from the origin:

$$R = \sqrt{P^2 + S^2}, \tag{B.2}$$

where P and S are the productivity and susceptibility scores, respectively. Risk ranges from 1.41, when all scores are equal to 1, to 4.24 when all scores are equal to 3. Assuming that all scores are equally likely, a third of the values are lower than 2.64 and a third of the values are above 3.18 (Hobday et al., 2007). Thus, the species with risks less than 2.64 are classified as at low risk, those between 2.64 and 3.18 as at medium risk, and those higher than 3.18 as at high risk (Hobday et al., 2007).

B.2.2.4 Data quality index

The data quality index was used to measure the uncertainty of the potential risk score due to the quality of the data used or to the total lack of data (Patrick et al., 2009). The index was based on five tiers, ranking from best data (or high belief in the score) to no data (or little belief in the score) as proposed in Patrick et al. (2009) (Table B.4). First, a score depending on the data quality was assigned to each productivity and level 2 susceptibility attribute. Level 1 susceptibility attributes data quality index were calculated as a weighted average of level 2 susceptibility attributes. Then, the overall productivity, susceptibility and risk data quality index were calculated as a weighted average of those main attributes. When there were no data for a particular attribute, it received the worse data quality score (i.e., "5") and it did not contribute to the productivity or susceptibility score. The data quality score and source of each productivity and susceptibility attributes are in Table C.6 and Table C.7 in Appendix C. The data quality index was divided into three categories as done by Patrick et al. (2009); poor >3.5, moderate 2.0–3.5; and good <2.0, in order to help the interpretation of the overall potential risk score uncertainty.

Table B.4: The five tiers of data quality used when evaluating the productivity and susceptibility of an individual species. Adapted from Patrick et al. (2009).

Tier	Description	Example
1	**Best data.** Information is based on collected data for the species and area of interest that is established and substantial.	Data-rich stock assessment, published literature documenting methods used.
2	**Adequate Data.** Information is based on limited coverage and corroboration, or for some other reason is deemed not as reliable as Tier 1 data.	Limited temporal or spatial data, relatively old information, etc.
3	**Limited Data.** Estimates with high variation and limited confidence and may be based on similar taxa or life-history strategy.	Similar genus or family, etc.
4	**Very Limited Data.** Information is based on expert opinion or on general literature reviews from wide range of species, or outside of region.	General data not referenced.
5	**No Data.** When there are no data on which to make even an expert opinion, this attribute data quality was scored as 5 and not provide a productivity or susceptibility score. When plotted, the susceptibility or productivity index score will be based on one less attribute and will be highlighted as such by its related quality score in order to highlight the uncertainty.	

B.2.2.5 Productivity attributes redundancy

Given that some of the productivity attributes, such as maximum length, maximum age, length at maturity and age at maturity attributes might be correlated, a sensitivity analysis was performed to determine whether any of these attributes are redundant in the PSA. First, the correlation between pairs of productivity attributes was assessed using a linear regression (pairs of attributes with R2 >0.5 were considered highly correlated). Then, correlated attributes redundancy was analysed removing one of the pairs of highly correlated attribute from the calculation of the overall productivity score and data quality score. In addition, we checked each correlated attributes data quality. This analysis allows to find any evidence to remove or maintain a correlated attribute to better approximation of the overall productivity score. For example; if the data quality of one of the correlated attributes is poor, and the overall productivity score changes significantly, removing this attribute, the overall productivity score calculation would be more accurate.

B.2.2.6 Comparison with IUCN and ICES categories

The overall potential risk estimates were compared to the International Union for Conservation of Nature red list classification (IUCN, 2019) that considers the following risk of extinction categories: not evaluated (NE), data deficient (DD), least concern (LC), near threatened (NT), vulnerable (VU), endangered (EN), and critically endangered (CR). If available, the European category was used and global category otherwise. We also analysed the ICES category (ICES, 2012) which classifies stocks according to the availability of data (Table A.1).

B.2.3 Species ranking

The objective of the species prioritisation approach was to identify those species that are highly at risk, exploited and/or economically important for the fishery. The approach is based on the methodology proposed in Osio et al. (2015) where, the ranking of the species is the product of the equitable contribution of landing volume, mean price and potential risk score. We extended this methodology including discards because the amount of some species discards in this fishery is high. In our approach, the revenue (€), the catch volume (Kg), and the overall fishery potential risk score (PSA) were used to define the ranking. In addition, the way to relate variables to determine the final ranking position was modified from Osio et al. (2015). In our case study, the catches for a small set of species were disproportionately high compared to most of the species included in the analysis, which masked the variability in catches across the lower value species. Thus, if the ranking criterion was set as the product of the variables, with their values standardised between 0 and 1, as in Osio et al. (2015), the position of each species would be primarily determined by the risk level, except for the few cases where landing or discard levels were high. Thus, to balance the contribution from each variable to the ranking position, the species were ranked according to each variable (revenue, catch and risk) importance separately and their final ranking position was determined as a product of their position in each of these separate classifications. Thus, the species situated at the top position in the ranking, contributes more to the revenue and/or is higher exploited and/or is at higher risk. On the contrary, the species on the bottom position in the ranking contribute less to the revenue and/or is lower exploited and/or is at lower risk.

The total revenue was calculated by multiplying the mean price per kilogram and the landed kilograms. The mean price (€/kg) of each species was calculated from 2008 to 2015 from the EC Annual Economic Report data (STECF, 2017). Fleets' historical landings and discards were obtained from the AZTI fisheries database across the period 2003–2017. Landed data was based on the sum of port sampling information. Discarded data were obtained with the extrapolation of the observer data, taking into account the measured sample size and the overall fishery. Finally, we analysed the sensitivity of the ranking results to the time period used for revenue and catch volume calculations (i.e., data from the last 10, 5, and 3 years).

B.3 Results

B.3.1 Species selection

A total of 64 species contributed up to 95% if we analyse the total landings and discards separately by metier. Three species (Norway lobster, European plaice, and spurdog), for which the Basque demersal mixed-fisheries have quota, were not part of the 95% contribution but were included in the analysis, resulting in a total of 67 species (Table B.5).

Table B.5: The scientific and common name of the selected species.

Scientific name	**Common name**
Argentina silus	Greater argentine
Argentina sphyraena	Argentine
Callionymus lyra	Dragonet
Cancer pagurus	Brown crab
Capros aper	Boarfish
Cepola rubescens	Red bandfish
Chelidonichthys cuculus	Red gurnard
Chelidonichthys lucerna	Tub gurnard
Chelidonichthys obscurus	Longfin gurnard
Conger conger	European conger
Dicentrarchus labrax	Sea bass
Dicologlossa cuneata	Wedge sole
Dipturus batis	Common skate
Dipturus oxyrinchus	Longnose skate
Eledone cirrhosa	Curled octopus
Engraulis encrasicolus	European anchovy
Eutrigla gurnardus	Grey gurnard
Galeorhinus galeus	Tope shark
Helicolenus dactylopterus	Blackbelly rosefish
Illex coindetii	Broadtail shortfin
Lepidorhombus boscii	Four-spot megrim
Lepidorhombus whiffiagonis	Megrim
Lepidotrigla cavillone	Large-scaled gurnard
Leucoraja naevus	Cuckoo ray
Loligo forbesii	Veined squid
Loligo vulgaris	European squid
Lophius budegassa	Black anglerfish
Lophius piscatorius	White anglerfish
Melanogrammus aeglefinus	Haddock
Merlangius merlangus	Whiting
Merluccius merluccius	Hake
Microchirus variegatus	Thickback sole
Micromesistius poutassou	Blue whiting
Molva molva	Ling

Scientific name	**Common name**
Mugil cephalus	Flathead grey mullet
Mullus surmuletus	Red mullet
Mustelus asterias	Starry smooth-hound
Mustelus mustelus	Common smooth-hound
Nephrops norvegicus	Norway lobster
Octopus vulgaris	Common octopus
Parastichopus regalis	Royal cucumber
Pegusa lascaris	Sand sole
Phycis blennoides	Greater forkbeard
Pleuronectes platessa	European plaice
Pollachius pollachius	Pollack
Raja clavata	Thornback ray
Raja fullonica	Shagreen ray
Raja montagui	Spotted ray
Raja undulata	Undulate ray
Sardina pilchardus	European pilchard
Scomber colias	Atlantic chub mackerel
Scomber scombrus	Mackerel
Scyliorhinus canicula	Catshark
Sepia elegans	Elegant cuttlefish
Sepia officinalis	Cuttlefish
Sepia orbignyana	Pink cuttlefish
Solea solea	Sole
Squalus acanthias	Spurdog
Todarodes sagittatus	European flying squid
Todaropsis eblanae	Lesser flying squid
Trachinus draco	Greater weever
Trachurus mediterraneus	Mediterranean horse mackerel
Trachurus trachurus	Horse mackerel
Trigloporus lastoviza	Streaked gurnard
Trisopterus luscus	Pouting
Trisopterus minutus	Poor cod
Zeus faber	John dory

Individual contribution to the total landings and discards of most of the species selected were less than 5% by gear (Figure B.1). In PTB fleet, 88% of the landings corresponded to hake. In LLS fleet, 56% of the landings corresponded to European conger, which together with hake and ling, represented 89% of the landings. In OTB fleet, the species that contributed more than 5% to the total landings were hake, catshark, red mullet, horse mackerel, black anglerfish, pouting and mackerel. They represented 54% of the total OTB landings. Regarding discards, hake, horse mackerel, and mackerel contributed 67% and 64% in PTB and OTB fleets, respectively. In addition, considering the contributions of blue whiting in PTB fleet and catshark in OTB fleet, these four species contributed 84% and 76% to the historical discards, respectively (Figure B.1). See Figure C.1, Figure C.2, and Figure C.3 in Appendix C for the landing and discard species contribution per gear in the last 10, 5, and 3 years.

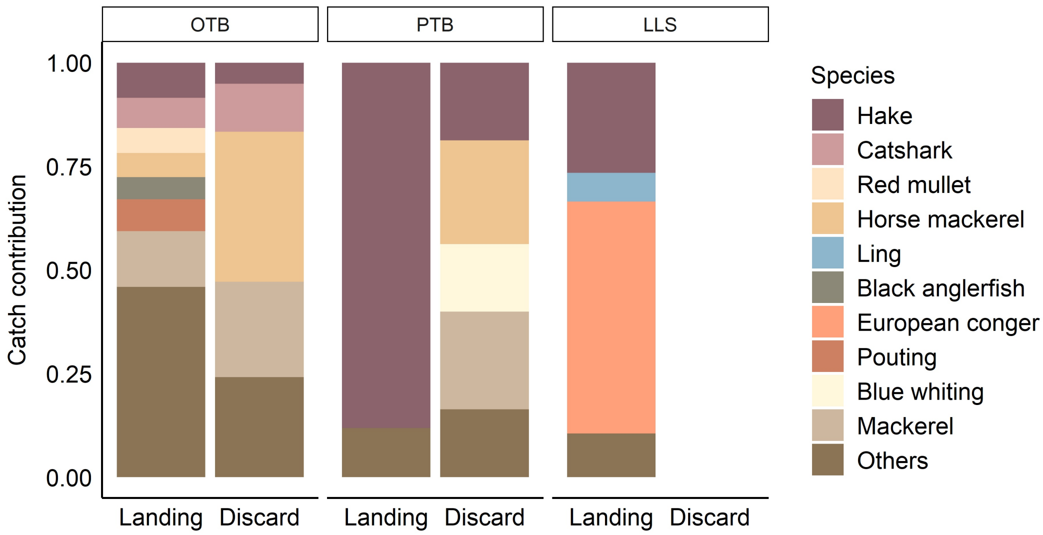


Figure B.1: Landings and discards species contribution (%) per gear: bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). Species whose individual contribution was less than 5% of the total were grouped in "Others".

B.3.2 Productivity-susceptibility analysis

B.3.2.1 Productivity

The two methods used to define the threshold of the productivity attributes, quantile and K-mean methods, showed similar thresholds for age at maturity attribute (Figure B.2). For other attributes, principally for fecundity, the thresholds are quite different, so the species were classified in different productivity category (low, medium, high) depending on the method used (Figure B.2). In fact, using K-mean method, due to the narrow range of most of the species attributes values, the thresholds only make differences between extreme values species masking the differences between most of the species (Figure B.2). We considered that the K-mean methodology created too coarse thresholds. Thus, we chose the quantile method to define those attributes thresholds in order to obtain higher differentiation between the productivity of all the species. See Table C.2 in Appendix C for productivity attribute values and the scores calculated using quantile and K-mean methods.

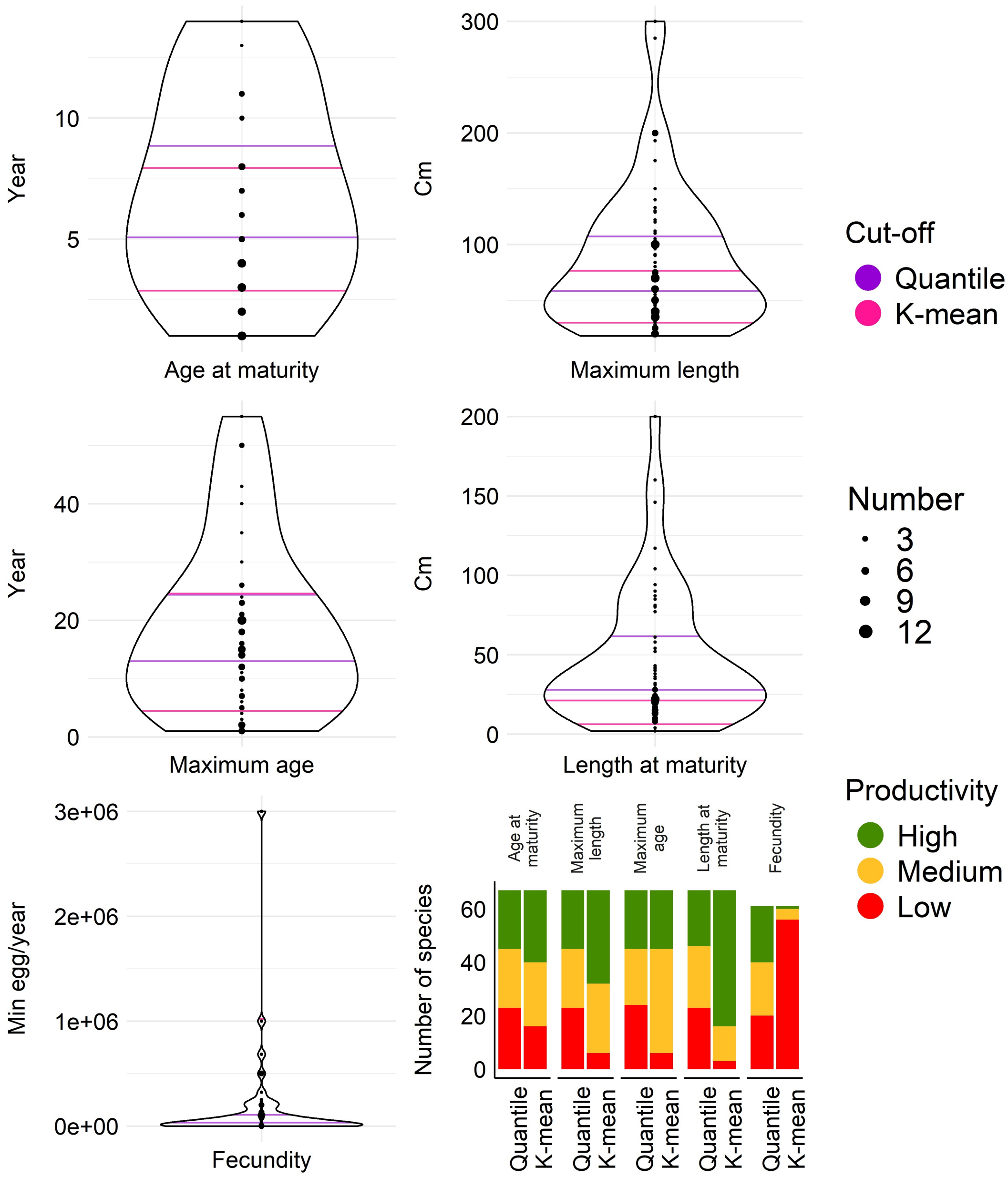


Figure B.2: Productivity attributes (age at maturity, maximum length, maximum age, length at maturity, and fecundity) threshold using different methods (Quantile and K-mean). Violin shape plots shows the species values distribution for each attribute, and the threshold depend on the method (Quantile and K-mean). Bar plot shows the number of species at each productivity level (High, Medium, Low) and attribute.

B.3.2.2 Single gear potential risk

There were no catches associated for two species in OTB fleet, for 13 species in PTB fleet, and for 43 species in LLS fleet. After removing in each fleet the species that had no catches associated, 30 species caught by OTB fleet were at high risk, 22 at medium risk, and 13 at low risk (Figure B.3). For PTB fleet, 19 species were at high risk, 25 at medium risk, and 10 at low risk (Figure B.3). In LLS fleet, 14 species were at high risk, 9 at medium risk, and 1 at low risk (Figure B.3). In the three gears, European conger, tope shark, blackbelly rosefish, black anglerfish, white anglerfish, hake, common smooth-hound, thornback ray, shagreen ray, and spurdog had high risk level (Table C.8 in Appendix C). See Table C.3, Table C.4, and Table C.5, and Table C.2 in Appendix C for values and scores used in the PSA.

B.3.2.3 Overall fishery potential risk

The number of species at high risk increased when the cumulative impact of all the fleets was assessed (Figure B.3). In AS cumulative scenario, 47 species were at high risk, 18 at medium risk, and 2 at low risk (Figure B.3). In particular, sharks and rays (common skate, longnose skate, tope shark, cuckoo ray, starry smooth-hound, common smooth-hound, thornback ray, shagreen ray, spotted ray, undulate ray, spurdog, and catshark) showed high risk values (Table C.8 in Appendix C).

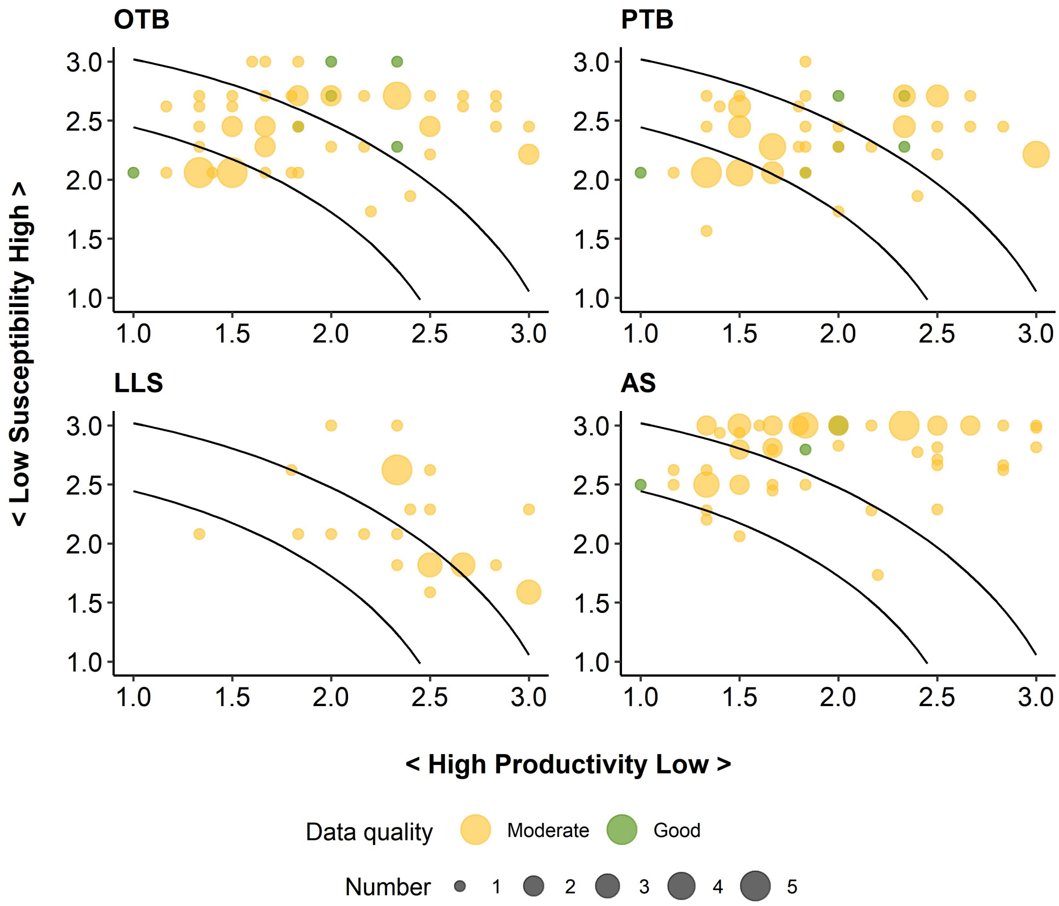


Figure B.3: Productivity-susceptibility analysis (PSA) plots for bottom otter trawlers (OTB), bottom pair trawlers (PTB), longliners (LLS) and overall fishery (AS). The curved lines divide the PSA plot into thirds, representing low, medium, and high risk, and group units of similar risk levels. Associated data quality category of each datum point was represented by colours and the size of the bullet represents the number of species with the same data quality, productivity and susceptibility values.

B.3.2.4 Data quality index

In all the species, the potential risk data quality score is in a moderate or good category (Figure B.3). In general, the data quality of the productivity attributes was good for most of the species. However, fecundity data was missing for boarfish, red bandfish, curled octopus, ling, flathead grey mullet and royal cucumber, and they received the worse data quality score (i.e., “5”) for this attribute (Table C.6 in Appendix C). The length at 50% selectivity data was missing for all the species in longliners, hence, the data quality score of this attribute was 5 for all the species (Table C.7 in Appendix C).

B.3.2.5 Productivity attributes redundancy

Two pairs of productivity attributes showed high correlation (R2 >0.5); age at maturity and length at maturity, and maximum length and length at maturity (Figure C.4 in Appendix C). Redundancy of those attributes was analysed removing one of the correlated two attributes and comparing the overall productivity risk score and data quality score obtained in each case (Figure B.4). In addition, as age at maturity and maximum length were linearly correlated with length at maturity, we also calculated the overall productivity risk score and data quality score removing age at maturity and maximum length attributes (Figure B.4). In all the cases, the obtained overall productivity risk scores were similar (Figure B.4). The main changes were in veined squid, royal cucumber, ling, broadtail shortfin, john dory, white anglerfish, sea bass and European conger, and minor changes in European anchovy, tope shark, common smooth-hound, sole, spurdog and horse mackerel (Figure B.4). In all the scenarios the data quality score of the overall productivity risk scores was good or moderate, except for longfin gurnard that was poor when the overall productivity risk score was calculated without maximum length attribute (Figure B.4). The correlated attributes data quality sores are similar (Table C.6 in Appendix C).

B.3.3 Species ranking

The final position of the species in the ranking was the product of their position in the revenue (€), catches volume (Kg) and risk (PSA score) rankings (Table B.6). Hake was in the first position of the final ranking. Although its price and discard volume were not high with respect to the rest of the species, it represented more than 50% of the total landing so it was at first position in the revenue and the catch rankings. In addition, it showed high risk level and it was situated at 9^{th} position in the risk ranking (Table B.6). Tope shark was at first position in the risk ranking but because of their low catch and revenue, it was at 12^{th} position in the final ranking. Black anglerfish, red mullet, European squid, white anglerfish, megrim and pouting were at 2^{nd}, 3^{rd}, 5^{th}, 6^{th}, 9^{th} and 10^{th} position in the final ranking respectively. They were situated at high position in the revenue ranking because of their high price and landing volume. Furthermore, their discards were very low (less than 0.1% each except in pouting that was around 0.3%) but due to their high landings volume, they were at high position in the catch ranking. They were all classified at high risk level, especially black and white anglerfish (Table B.6). Catshark, horse mackerel, and mackerel were situated at 4^{th}, 7^{th} and 8^{th} position in the final ranking. Because of their high discards volume, they were all situated at the top positions of the catch ranking. They were also among the most landed species but with low price (less than 1€/Kg), so they were situated above the middle of the revenue ranking. Catshark and horse mackerel were classified at high risk level and situated at 6^{th} and 15^{th}

position in the risk ranking respectively (Table B.6). Mackerel was at medium risk level and at 29th position in the risk ranking (Table B.6).

The ranking of the species was robust to recent changes in catch composition (Figure B.1, and Figure C.1, Figure C.2, and Figure C.3 in Appendix C). The top-ten ranked species positions remained practically unchanged in all the time periods analysed. The exceptions were red mullet, horse mackerel, mackerel and megrim. The positions of these four species varied between 5 and 10 positions in the rank. Common octopus was the most sensitive species to the time period considered in the analysed (Figure B.5). Considering only the most recent data (2008–2017, 2013–2017, 2015–2017), its position in the ranking decreased almost 30 ranks respect to the analysis including the whole time period (2003–2017).

Regarding the IUCN conservation status classification, all the species considered at extinction risk (VU, EN, NT, or CR) were at high risk according to PSA, except European pilchard, which was at medium risk level and is classified as NT by IUCN (Table B.6). The top-ten ranked species are classified at least concern category in the IUCN, except red mullet and European squid that are at data deficient category. Concerning ICES categories, 33 of the species included in this analysis were not assessed by ICES. Most of these species were situated below the tenth position in the general ranking, except European squid and Pouting (Table B.6). ICES (2012) identifies the species classified in ICES category 3 or above as data-limited species. The top-ten species are classified in ICES category 1 except catshark and red mullet, which are classified in ICES category 3 and 5 respectively.

Finally, only 27 of the species caught by the Basque demersal mixed-fisheries are included in the European TAC and quota system, and six of those species were in the top-ten positions (Table B.6).

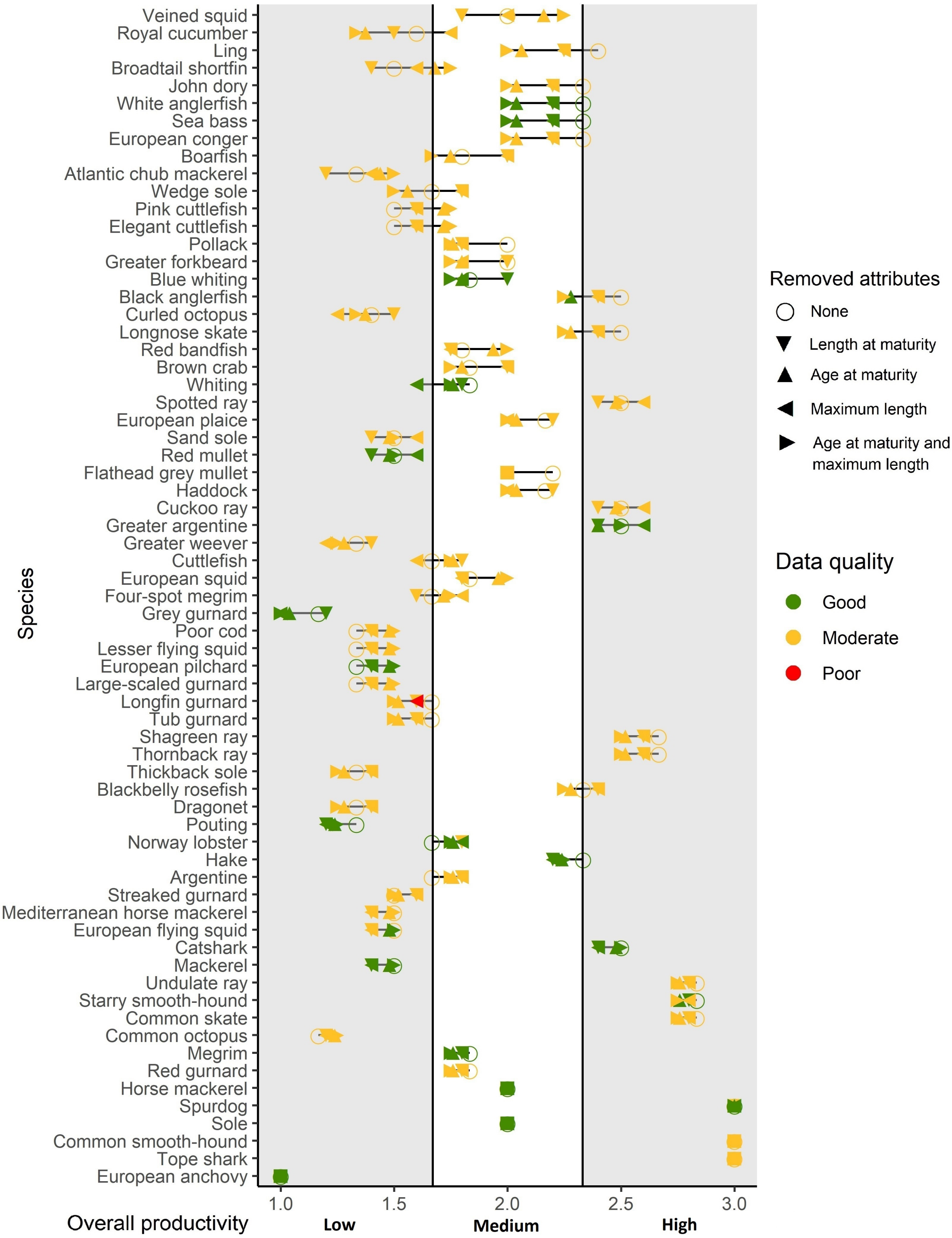


Figure B.4: Productivity attributes redundancy analysis removing linearly correlated attributes. Overall productivity score and level (Low, Medium, High) of each species using all the productivity attributes (age at maturity, maximum age, length at maturity, maximum length, fecundity, and reproductive strategy), removing length at maturity attribute, removing age at maturity attribute, removing maximum length attribute, and removing age at maturity and maximum length attributes. Species are ordered from high differences to not differences between overall productivity scores of different scenarios. Data quality levels (Good, Moderate, Poor) refer to the overall productivity of each scenario.

Table B.6: Summary of the final ranking. The final ranking position based on the product of revenue landing [(2003–2017 Kg) * price (€/kg)], catch [2003–2017 Kg] and potential risk score [overall fishery risk] ranking positions (R). Absolute and relative (percentage respect to the total analysed period) values of the revenue, catch (landings L, plus discards D) and the risk score and level (low, medium, and high) were showed to allow a better interpretation of the ranking position. ICES categories (I) based on the data availability [from 1 to 6, and not assessed (N)], IUCN conservation status (IU) [not evaluated (NE), data deficient (DD), least concern (LC), near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR)] and if the species are subject to catch restrictions for the Basque demersal mixed-fisheries (C) information. IUCN* means that this category refers to the global evaluation (not to the European evaluation like the rest). Y* means that there is no specific TAC for this species. Those species fishing opportunities are managed through an overall TAC by management unit, which includes almost all the species of skates and rays.

Species	Final Ranking										Data						Information		
		Revenue			Catch			Risk											
		R	€	%	**R**	Kg	%	**R**	S	L	€/Kg	Land (Kg)	Land (%)	Disc (kg)	Disc (%)	I	IU	C
Hake	1	1	3E+8	50.73	1	8E+7	36.21	9	3.80	H	3.33	8E+7	53.26	5E+6	6.24	1	LC	Y
Black anglerfish	2	6	1E+7	2.87	8	4E+6	1.65	6	3.91	H	3.95	4E+6	2.55	6E+4	0.08	1	LC	Y
Red mullet	3	2	4E+7	6.98	7	4E+6	1.80	22	3.35	H	8.67	4E+6	2.82	4E+3	0.00	5	DD	N
Catshark	4	18	5E+6	0.87	4	1E+7	5.48	6	3.91	H	0.94	5E+6	3.24	8E+6	9.34	3	LC	N
European squid	5	3	4E+7	6.82	10	4E+6	1.52	16	3.52	H	10.02	4E+6	2.38	4E+3	0.00	N	DD*	N
White anglerfish	6	5	2E+7	3.42	11	3E+6	1.32	9	3.80	H	5.9	3E+6	2.03	7E+4	0.08	1	LC	Y
Horse mackerel	7	21	3E+6	0.63	2	4E+7	15.10	15	3.61	H	0.83	4E+6	2.64	3E+7	36.73	1	LC	Y
Mackerel	8	9	1E+7	2.14	3	3E+7	12.88	29	3.17	M	1.05	1E+7	7.15	2E+7	22.79	1	LC	Y
Megrim	9	4	2E+7	3.73	16	2E+6	0.97	23	3.34	H	8.85	2E+6	1.47	7E+4	0.08	1	LC	Y
Pouting	10	10	1E+7	1.88	6	6E+6	2.40	25	3.28	H	1.82	5E+6	3.60	3E+5	0.29	N	LC	N
Sea bass	11	8	1E+7	2.61	25	1E+6	0.51	9	3.80	H	11.4	1E+6	0.80	0.00	0.00	1	LC	N
Thornback ray	11	20	3E+6	0.64	18	2E+6	0.84	5	4.01	H	1.8	2E+6	1.24	1E+5	0.14	3	NT	Y
Tope shark	12	40	3E+5	0.05	50	1E+5	0.05	1	4.24	H	2.14	1E+5	0.08	0.00	0.00	5	VU	Y*
European conger	13	17	5E+6	0.91	15	3E+6	1.10	9	3.80	H	1.89	2E+6	1.69	5E+4	0.06	N	LC	N
John dory	14	13	7E+6	1.36	26	1E+6	0.49	9	3.80	H	6.32	1E+6	0.75	2E+4	0.03	N	DD	N
Cuttlefish	15	7	1E+7	2.66	14	3E+6	1.19	32	3.00	M	4.99	3E+6	1.86	1E+4	0.02	N	LC*	N
Argentine	16	11	9E+6	1.69	13	3E+6	1.21	26	3.27	H	4.48	2E+6	1.32	9E+5	1.02	N	LC	N
Common smooth-hound	17	28	2E+6	0.30	37	4E+5	0.16	4	4.11	H	4.12	4E+5	0.25	2E+3	0.00	3	VU	Y*
Tub gurnard	18	12	7E+6	1.44	19	2E+6	0.73	19	3.43	H	4.44	2E+6	1.13	2E+4	0.03	N	LC	N

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	Final Ranking																			
		Revenue			**Catch**			**Risk**			**Data**					**Information**				
Species		**R**	€	%	**R**	Kg	%	**R**	S	L	Price	Land (Kg)	Land (%)	Disc (kg)	Disc (%)	I	IU	C		
Spurdog	19	47	7E+4	0.01	54	6E+4	0.03	2	4.23	H	1.57	4E+4	0.03	2E+4	0.03	1	EN	Y		
Blue whiting	20	33	8E+5	0.16	5	8E+6	3.31	31	3.10	M	1.36	6E+5	0.40	7E+6	8.36	1	LC	Y		
Starry smooth-hound	21	39	3E+5	0.06	48	2E+5	0.08	3	4.13	H	2.47	1E+5	0.09	4E+4	0.05	3	NT	Y*		
Whiting	22	22	3E+6	0.49	17	2E+6	0.93	16	3.52	H	1.33	2E+6	1.28	3E+5	0.32	5	LC	Y		
Cuckoo ray	23	29	1E+6	0.21	22	1E+6	0.53	10	3.76	H	1.43	8E+5	0.51	5E+5	0.58	3	LC	Y		
Poor cod	24	19	4E+6	0.76	12	3E+6	1.21	37	2.83	M	2.03	2E+6	1.31	9E+5	1.04	N	LC	N		
Sole	25	15	2E+6	1.02	39	4E+5	0.16	15	3.61	H	15.76	3E+5	0.23	3E+4	0.03	1	LC	Y		
Grey gurnard	25	25	5E+6	0.42	9	4E+6	1.62	39	2.76	M	2.04	1E+6	0.71	3E+6	3.18	N	LC	N		
Four-spot megrim	26	14	6E+6	1.18	30	8E+5	0.34	27	3.25	H	7.97	8E+5	0.52	2E+4	0.03	5	LC	Y		
Common octopus	27	16	5E+6	0.97	21	1E+6	0.54	36	2.87	M	4.46	1E+6	0.76	1E+5	0.14	N	LC*	N		
Broadtail shortfin	28	26	2E+6	0.38	24	1E+6	0.51	22	3.35	H	2.09	9E+5	0.63	3E+5	0.31	N	LC	N		
Curled octopus	29	23	2E+6	0.48	23	1E+6	0.53	28	3.25	H	2.08	1E+6	0.81	4E+4	0.04	N	LC*	N		
Shagreen ray	30	57	2E+3	0.00	64	1E+3	0.00	5	4.01	H	2.02	1E+3	0.00	0.00	0.00	6	VU	Y*		
Blackbelly rosefish	31	42	2E+5	0.04	53	7E+4	0.03	9	3.80	H	3.68	6E+4	0.04	1E+4	0.01	N	LC	N		
Ling	32	37	5E+5	0.10	42	3E+5	0.11	13	3.67	H	1.95	3E+5	0.18	8E+2	0.00	3	LC	Y		
Red gurnard	33	49	4E+4	0.01	28	1E+6	0.41	16	3.52	H	2.17	2E+4	0.01	9E+5	1.10	6	LC	N		
Brown crab	34	43	2E+5	0.04	32	6E+5	0.26	16	3.52	H	2.29	8E+4	0.05	5E+5	0.62	N	NE	N		
Undulate ray	35	54	7E+3	0.00	61	1E+4	0.00	7	3.89	H	2.02	3E+3	0.00	7E+3	0.01	6	NT	Y		
Greater forkbeard	36	36	6E+5	0.11	44	2E+5	0.10	15	3.61	H	3.2	2E+5	0.12	4E+4	0.04	3	DD	N		
Red bandfish	37	38	5E+5	0.09	43	3E+5	0.11	17	3.50	H	2.06	2E+5	0.16	3E+4	0.03	N	LC	N		
Wedge sole	38	24	2E+6	0.48	36	4E+5	0.18	33	2.96	M	7.33	3E+5	0.23	8E+4	0.09	N	LC	N		
Boarfish	39	60	0.00	0.00	29	9E+5	0.40	17	3.50	H	0.85	0.00	0.00	9E+5	1.09	3	LC	N		
Elegant cuttlefish	40	27	2E+6	0.31	46	2E+5	0.08	24	3.30	H	8.6	2E+5	0.13	5E+3	0.01	N	DD*	N		
Pollack	41	34	6E+5	0.12	49	1E+5	0.06	18	3.46	H	4.66	1E+5	0.09	2E+2	0.00	5	LC	Y		
Lesser flying squid	42	35	6E+5	0.12	35	4E+5	0.18	25	3.28	H	2.07	3E+5	0.20	1E+5	0.14	N	LC*	N		
Haddock	43	55	5E+3	0.00	52	1E+5	0.04	11	3.70	H	1.4	4E+3	0.00	1E+5	0.12	N	LC	N		

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Table B.6 – Continued from previous page

Species	Final Ranking	Revenue R	Revenue €	Revenue %	Catch R	Catch Kg	Catch %	Risk R	Risk S	Risk L	Data Price	Data Land (Kg)	Data Land (%)	Data Disc (kg)	Data Disc (%)	Information I	Information IU	Information C
Common skate	44	59	1E+1	0.00	67	9E+0	0.00	8	3.86	H	1.15	9E+0	0.00	0.00	0.00	6	CR	Y*
Longfin gurnard	45	41	2E+5	0.04	33	5E+5	0.22	26	3.27	H	1.22	2E+5	0.13	3E+5	0.37	N	LC	N
Veined squid	46	45	1E+5	0.03	57	3E+4	0.01	15	3.61	H	5.65	2E+4	0.02	2E+3	0.00	N	LC*	N
Greater argentine	47	60	0.00	0.00	56	4E+4	0.02	12	3.69	H	4.57	0.00	0.00	4E+4	0.05	3	LC	N
Spotted ray	48	51	2E+4	0.00	58	2E+4	0.01	14	3.65	H	1.96	1E+4	0.01	9E+3	0.01	3	LC	Y*
Greater weever	49	32	9E+5	0.17	31	8E+5	0.33	42	2.57	L	1.86	5E+5	0.32	3E+5	0.36	N	LC	N
Dragonet	50	60	0.00	0.00	20	2E+6	0.66	37	2.83	M	0.61	0.00	0.00	2E+6	1.81	N	LC	N
Norway lobster	51	46	1E+5	0.02	59	1E+4	0.01	19	3.43	H	17.49	7E+3	0.00	5E+3	0.01	1	LC*	Y
Mediterranean horse mackerel	52	44	2E+5	0.03	41	3E+5	0.13	29	3.17	M	0.82	2E+5	0.14	1E+5	0.13	N	LC	Y
Royal cucumber	53	60	0.00	0.00	45	2E+5	0.09	20	3.40	H	55.47	0.00	0.00	2E+5	0.25	N	LC*	N
Pink cuttlefish	54	31	1E+6	0.18	51	1E+5	0.05	35	2.91	M	9.35	1E+5	0.07	1E+4	0.01	N	DD*	N
Thickback sole	55	30	1E+6	0.20	47	2E+5	0.08	41	2.64	M	6.29	2E+5	0.11	3E+4	0.03	N	LC	N
European pilchard	56	60	0.00	0.00	27	1E+6	0.47	37	2.83	M	1.25	0.00	0.00	1E+6	1.28	2	NT	N
European flying squid	57	50	3E+4	0.01	60	1E+4	0.01	22	3.35	H	2.68	1E+4	0.01	1E+3	0.00	N	LC*	N
Atlantic chub mackerel	58	53	1E+4	0.00	34	5E+5	0.19	37	2.83	M	0.6	2E+4	0.02	4E+5	0.50	N	LC	N
Flathead grey mullet	59	48	6E+4	0.01	38	4E+5	0.16	38	2.80	M	1.21	5E+4	0.03	3E+5	0.39	N	LC*	N
Longnose skate	60	58	8E+1	0.00	66	4E+1	0.00	21	3.39	H	1.83	4E+1	0.00	0.00	0.00	6	NT	Y*
European anchovy	61	60	0.00	0.00	40	4E+5	0.16	40	2.69	M	2.05	0.00	0.00	4E+5	0.43	1	LC	Y
Streaked gurnard	62	52	2E+4	0.00	55	6E+4	0.03	35	2.91	M	1.55	1E+4	0.01	5E+4	0.06	N	DD	N
European plaice	63	60	0.00	0.00	62	1E+4	0.00	30	3.14	M	6.71	0.00	0.00	1E+4	0.01	5	LC*	Y
Large-scaled gurnard	64	60	0.00	0.00	63	1E+3	0.00	34	2.94	M	2.55	0.00	0.00	1E+3	0.00	N	LC	N
Sand sole	65	56	3E+3	0.00	65	4E+2	0.00	43	2.55	L	7.31	4E+2	0.00	0.00	0.00	N	LC	N

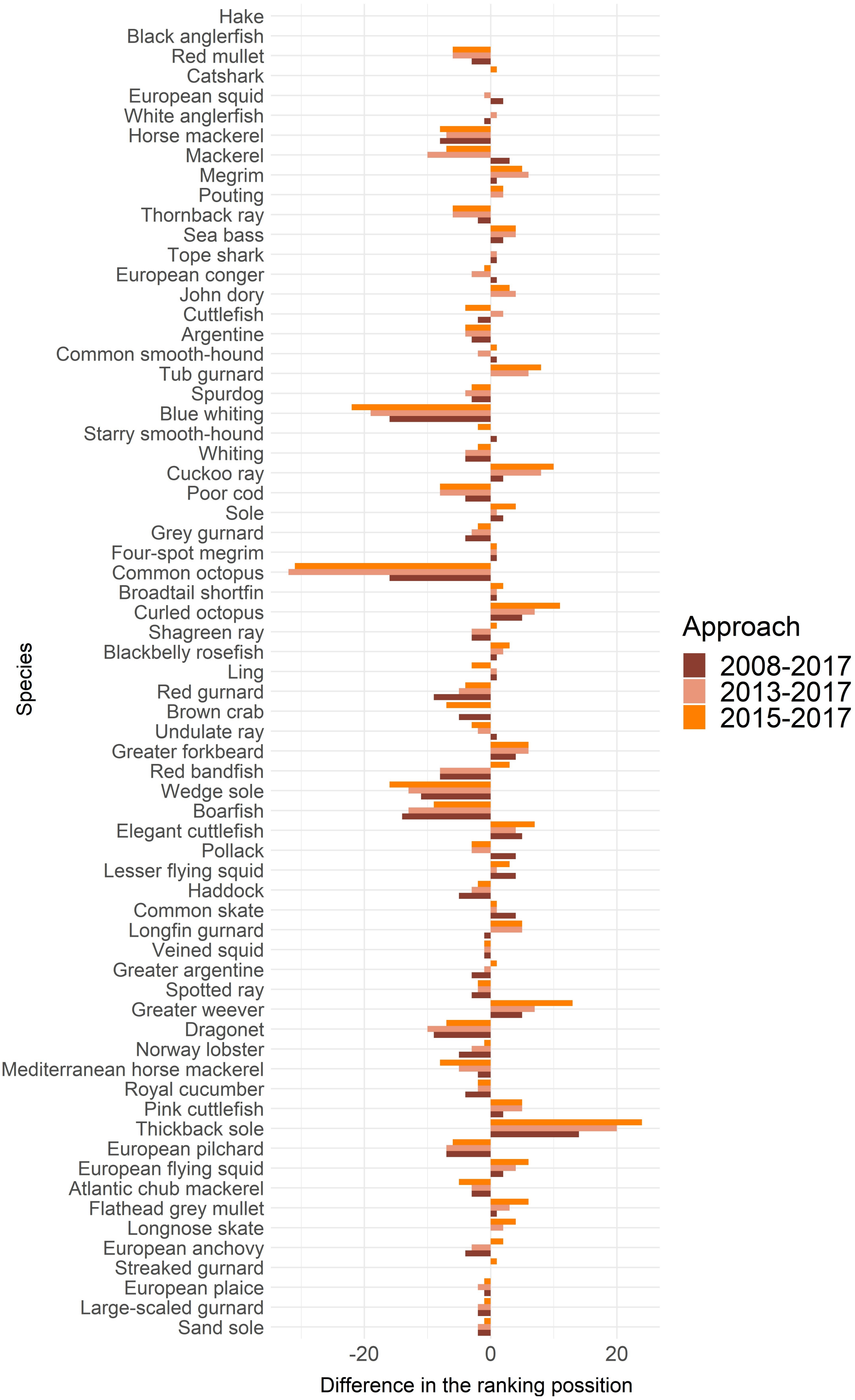


Figure B.5: Ranking robustness to the landing and discards time period data. Reference landing and discard time period: 2003–2017.

B.4 Discussion

Decision makers require robust scientific advice to focus research and management efforts on the species that need them the most (Smith et al., 2009). In this chapter, we proposed an approach to respond to this challenge identifying those species with potentially high risk for the effects of fishing, economically great importance for the fishery and/or highly exploited by the fisheries. The species ranking method proposed in this work was an extension of the method proposed by Osio et al. (2015) to include discards. The approach was applied to the Basque demersal mixed-fisheries. We ranked the species exploited by the Basque demersal mixed-fisheries based on their potential risk delivered from a PSA, their contribution to the revenue of the fishery, and their catch volume. The potential risk accounted for the overall Basque demersal mixed-fisheries based on the aggregated susceptibility of different fleets proposed by Micheli et al. (2014).

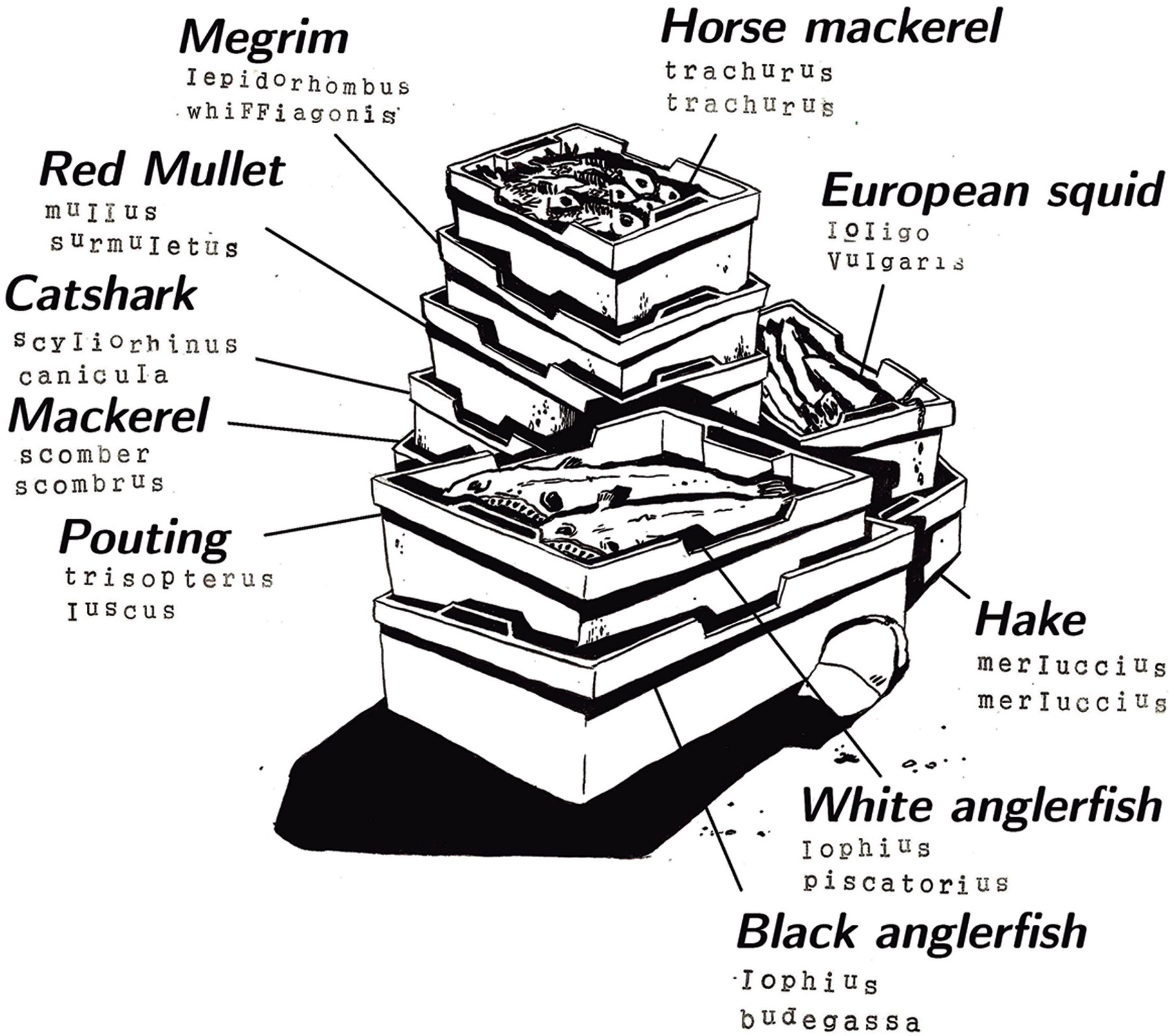
Despite the extensive use of PSA, there is no unified recommended framework for its application. The method is tailored to each case, and researchers use different methodologies to calculate potential risk values (Hordyk and Carruthers, 2018). Our study was based in the PSA attributes proposed by Hobday et al. (2011), which scores seven productivity and four susceptibility attributes to calculate the species potential risk. Despite the trophic level attribute, the other six productivity attributes proposed by Hobday et al. (2011) were included in this analysis. The trophic level attribute was excluded because it is not a useful productivity measure for all the species analysed. Age at maturity, maximum length and length at maturity attributes were highly correlated but the differences in the overall productivity risk score and data quality score calculated in each redundancy analysis scenario were small. Thus, we have not found any evidence to remove and/or choose one of those correlated attributes and we introduced all of them in the PSA analysis. Some authors confirmed that the inclusion of more attributes allows more comprehensive assessment of productivity (Patrick et al., 2009). There are other extended PSA methods that includes more attributes (Patrick et al., 2010) than used in this work but these attributes are often correlated and difficult to measure (e.g., mortality) (Duffy and Griffiths, 2017) so it could be assumed that the data quality of those correlated attributes are worse and consequently redundant. Some researches have excluded fecundity (along with other reproductive attributed) from the PSA analysis (Hordyk and Carruthers, 2018) due to the lack of correlation between fecundity, early life-history survival and recruitment. However, it is important to note that in this PSA study, we assessed the potential reproductive capacity of the species measuring the species productivity in terms of the number of eggs spawned. The assessment does not directly measure recruitment, which is influenced by various factors such as environmental conditions, predation, and others. The method to set the thresholds in the productivity attributes, differs between authors. In this work, two of the most used methods, the quantile and K-mean method, were analysed. In this case, the species were very diverse, from cephalopods to sharks, so their values varied in a wide range inside each attribute, although, most of the species values varied in a narrow range of the attribute values distribution. Consequently, quantile method was more useful than K-mean method to find productivity differences between all the species. In fact, K-mean methodology created too coarse thresholds, highlighting only species with extreme values and grouping most of the species at the same productivity category. As an alternative to the scoring system, some PSA approaches use mathematical equations to calculate productivity and susceptibility scores (Kirby, 2006). Those methods application often is limited for unknown data, for example the PSA approach proposed by Kirby (2006) could not be applied without mean length of catch data. Here, we can not calculate susceptibility of all the analysed species with this approach, because this data for most of the species was unknown. The way to address attributes when data is missing also differs between PSA approaches (Hobday et al., 2011; ICES, 2013; Patrick et al., 2010; Smith et al., 2009). Some authors following the precautionary approach, use the highest score value to the missing attribute (Hobday et al., 2011; ICES, 2013). This approach has been criticized for overestimating risk values (false positive) (Hordyk and Carruthers, 2018; Zhou et al., 2016). In this analysis, the missing attributes were accounted using the data quality index (Lucena-Fredou et al., 2017; Patrick et al., 2009). Except the lack of fecundity data for some species and longliners selectivity data for all the species, the quality index for all the species was moderate or good.

Although the PSA results have been compared with other approaches (ICES, 2013; Lucena-Fredou et al., 2017; Zhou et al., 2016), the method was tested quantitatively for the first time by Hordyk and Carruthers (2018). They found that under the most favourable species status conditions, the expected success of PSA in categorising risk is approximately 66%, and less than 50% under certain conditions such as high exploitation rate and low initial population size. They affirmed that the information required to develop a quantitative risk assessment is comparable to that needed to conduct a PSA. Thus, Hordyk and Carruthers (2018) advocated the development of quantitative risk assessment approaches such as simulation modelling to identify species at a greater risk of overexploitation. However, in practice, the use of quantitative risk assessment approaches to model a great number of data-limited species and fisheries incorporating uncertainty can be a difficult and time-consuming task (Bentley, 2014). In addition, data-limited species are often low in value, so the time and resources available to assess these species are also constrained (Bentley, 2014). Furthermore, data used in PSA approach is not always enough to apply simulation approaches like the one used by Hordyk and Carruthers (2018). PSA methods have demonstrated their value as a first quick approach to provide a screen of relative potential risk of data-limited and data-rich species within the same framework (Ormseth and Spencer, 2011). It has been considered a valuable tool to identify species most at risk in a data-limited situation by ICES (2012) and MSC (2001). In fact, although the results of the PSA from individual assessments should be interpreted with caution, the relative comparisons of risk among species are robust (Micheli et al., 2014). In the evaluation of management plans is a useful semi-quantitative approach to screen the species that should be included in the simulation models.

The species prioritisation approach presented in this chapter, allows ranking species from those most at risk, most exploited and/or economically most important, to the least. The contribution to the ranking of each of those three variables were balanced and the method was robust for the time period analysed. We used the catch variable to measure the fishing activity pressure, implicitly assuming that all the species retained die. However, we should have had taken into account the high post-capture survival of sharks and rays. An alternative approach is to introduce a proxy of the relative abundances of the species instead of catches (Arrizabalaga et al., 2011). However, most of the species in this case study were data-limited and their abundance was unknown.

Given the big amount of species caught by mixed-fisheries, is practically impossible to include all of them in the simulation model. Hence, approaches that consider life-history of the species and the activity of the fleets to select the species that should be included in the simulation, like the one presented here, are useful screening tools. We recommend to include the species situated at the top of the ranking ensuring that they represent a big proportion of the total catch and revenue of the Basque demersal mixed-fisheries and that the risk associated with the remaining species is not likely to be enhanced by the activity of the fleet. In this case, including the top-ten ranked species, an adequate evaluation of the management plans for the Basque demersal mixed-fisheries is ensured. The top-ten ranked species represented more than 76% of the total Basque demersal income, landings, and discards volume. Those species (except mackerel) are potentially at high risk. Especially, more effort should be done to advance in the correct management of red mullet, catshark and European squid. They are in the third, fourth and fifth position in the general ranking. However, their exploitation is not subject to any catch restriction, they are classified in a data-limited ICES category, and they show high potential risk to the fishery. In addition, the relative catch contribution to the fishery of red mullet, was reduced in the last years. Under the landing obligation, principally sharks and rays will also need more attention since they will be potential choke species due to their low TAC.

The model was only applied to the Basque demersal mixed-fisheries, because we had complete historical catch data at stock and fleet levels for that fishery, but not for the whole Spanish and French demersal fishery operating in the Bay of Biscay. The main part of the Spanish demersal fishery that operates in the Northern part of the Bay of Biscay (ICES 8abd) is Basque, so most of the Spanish fleet was covered by the analysis. The ranking of the species obtained in this chapter was presented in the ICES working group of the mixed-fisheries methods in 2018 concluding that the ranking was adequate for the whole demersal mixed-fisheries operating in the Bay of Biscay because most of the important species in terms of their contribution to the revenue of the French and Spanish fleets were in the top of the ranking.



Appendix C

Supplementary material to Chapter 2

This supplementary material comprises each gear adult habitat and bathymetry susceptibility attributes thresholds scores (Section C.1), productivity attributes values and scores using quantile and k-mean methods (Section C.2), each gear susceptibility attributes values and scores, and presence of each species in each gear historical catches (Section C.3), data quality score and source of each productivity and susceptibility attributes (Section C.4), potential risk scores and their corresponding data quality index (Section C.5), landing and discard species contribution per gear in the last 10, 5, and 3 years (Section C.6), and linear regressions for pair of productivity attributes (Section C.7).

C.1 Each gear specific adult habitat and bathymetry susceptibility attributes thresholds scores

Table C.1: Adult habitat and bathymetry susceptibility attributes thresholds scores for each gear. Gears: bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). Risk level: H=High, M=Medium, and L=Low.

		Code	LLS	OTB	PTB
Adult Habitat	Soft Bottom; sand, mud	SB	H	H	H
	Hard Bottom; rocky, reefs	HB	M	L	L
	Surface (pelagic or epipelagic)	EP	M	M	M
	Midwater	MP	M	M	M
Bathymetry	0-10	1	L	L	L
	10-24	2	L	H	L
	24-300	3	L	H	H
	300-400	4	H	H	H
	400-1400	5	H	H	L
	>1400	6	H	L	L

C.2 Productivity attributes values and scores

Table C.2: Productivity attributes values and scores using quantile (Q) and k-mean (K) methods. Age at maturity, Maximum age, Length at maturity, Maximum length, Fecundity, and Reproductive strategy. Reproductive strategy: fish which release their gametes into the water (Broadcast spawners – External fertilization, BS), species that lay eggs (oviparity, egg layers – Internal fertilization, DS), and ovoviviparity and viviparity species (Live bearers – Internal fertilization, LB)

	Age at mat			Max age			Length at mat			Max length			Fecundity			Rep. strategy		Total score	
Species	Year	Q	K	Year	Q	K	Cm	Q	K	Cm	Q	K	egg/yr	Q	K	Code	Score	Q	K
Greater argentine	8.2	3	3	35	3	3	38	3	1	70	3	2	4478	3	3	BS	1	2.50	2.17
Argentine	2.5	2	1	16	2	2	12.5	1	1	35	1	1	4478	3	3	BS	1	1.67	1.50
Dragonet	3.5	2	2	7	1	1	17.4	1	1	30.5	1	1	5000	2	3	BS	1	1.33	1.50
Brown crab	8	3	3	50	3	3	13.2	1	1	20	1	1	250000	1	3	DS	2	1.83	2.17
Boarfish	5	3	2	26	3	2	8.05	1	1	18	1	1				BS	1	1.80	1.40
Red bandfish	1.5	1	1	30	3	2	21.9	2	1	80	2	2				BS	1	1.80	1.40
Red gurnard	3.7	2	2	21	3	2	26.6	2	1	50	2	1	100000	1	3	BS	1	1.83	1.67
Tub gurnard	3	2	2	14	2	2	21.6	2	1	75	2	2	100000	1	3	BS	1	1.67	1.83
Longfin gurnard	3	2	2	14	2	2	21.6	2	1	50.5	2	1	100000	1	3	BS	1	1.67	1.67
European conger	10	3	3	20	3	2	200	3	3	300	3	3	3000000	1	1	BS	1	2.33	2.17
Sea bass	7	3	3	20	3	2	42.1	3	1	96	3	2	231875	1	3	BS	1	2.33	2.00
Wedge sole	6	3	2	23	3	2	17.1	1	1	30	1	1	160000	1	3	BS	1	1.67	1.67
Common skate	11	3	3	23	3	2	160	3	3	285	3	3	40	3	3	DS	2	2.83	2.67
Longnose skate	7	3	3	10	1	1	104	3	2	150	3	2	40	3	3	DS	2	2.50	2.17
Curled octopus	1.5	1	1	3	1	1	11.1	1	1	55	1	1				DS	2	1.40	1.20
European anchovy	1	1	1	5	1	1	9.7	1	1	20	1	1	110000	1	3	BS	1	1.00	1.33
Grey gurnard	2	1	1	8	1	1	18.8	1	1	50	1	1	200000	1	3	BS	1	1.17	1.33

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	Age at mat			Max age			Length at mat			Max length			Fecundity			Rep. strategy		Total score	
Species	Year	Q	K	Year	Q	K	Cm	Q	K	Cm	Q	K	egg/yr	Q	K	Code	Score	Q	K
Tope shark	10	3	3	55	3	3	146	3	3	193	3	3	6	3	3	LB	3	3.00	3.00
Blackbelly rosefish	13	3	3	43	3	3	32	2	1	47	2	1	31767	2	3	DS	2	2.33	2.17
Broadtail shortfin	0.6	1	1	0.9	1	1	20.7	2	1	32	2	1	46500	2	3	DS	2	1.50	1.50
Four-spot megrim	2.5	2	1	15	2	2	20	2	1	40	2	1	42000	2	3	BS	1	1.67	1.50
Megrim	2.5	2	1	15	2	2	20	2	1	60	2	1	26522	2	3	BS	1	1.83	1.50
Large-scaled gurnard	2	1	1	7	1	1	8	1	1	20	1	1	503	3	3	BS	1	1.33	1.33
Cuckoo ray	8	3	3	12	2	2	57.5	3	2	71	3	2	60	3	3	DS	2	2.50	2.33
Veined squid	1.2	1	1	1.5	1	1	40	3	1	70	3	2	1300	3	3	DS	2	2.00	1.67
European squid	0.8	1	1	1	1	1	28	2	1	48	2	1	1300	3	3	DS	2	1.83	1.50
Black anglerfish	8.2	3	3	21	3	2	53.6	3	2	175	3	3	87569	2	3	BS	1	2.50	2.33
White anglerfish	14	3	3	20	3	2	93.9	3	2	200	3	3	1000000	1	2	BS	1	2.33	2.17
Haddock	3.5	2	2	20	3	2	34.9	2	1	112	2	2	55000	2	3	BS	1	2.17	1.83
Whiting	1.5	1	1	20	3	2	27.8	2	1	91	2	2	200000	1	3	BS	1	1.83	1.67
Hake	4	2	2	20	3	2	42.8	3	1	140	3	2	52000	2	3	BS	1	2.33	1.83
Thickback sole	3	2	2	14	2	2	9	1	1	35	1	1	500000	1	2	BS	1	1.33	1.50
Blue whiting	3	2	2	20	3	2	15	1	1	50	1	1	6000	2	3	BS	1	1.83	1.67
Ling	5.5	3	2	14	2	2	90	3	2	200	3	3				BS	1	2.40	2.00
Flathead grey mullet	3.5	2	2	16	2	2	35.4	3	1	100	3	2				BS	1	2.20	1.60
Red mullet	3	2	2	11	1	1	22	2	1	44.5	2	1	19640	2	3	BS	1	1.50	1.50
Starry smooth-hound	6	3	2	18	2	2	87	3	2	133	3	2	7	3	3	LB	3	2.83	2.33
Common smooth-hound	11	3	3	24	3	2	117	3	2	129	3	2	4	3	3	LB	3	3.00	2.50

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	Age at mat			Max age			Length at mat			Max length			Fecundity			Rep. strategy		Total score	
Species	Year	Q	K	Year	Q	K	Cm	Q	K	Cm	Q	K	egg/yr	Q	K	Code	Score	Q	K
Norway lobster	2.5	2	1	10	1	1	2.23	1	1	24	1	1	1000	3	3	DS	2	1.67	1.50
Common octopus	1.5	1	1	2	1	1	10	1	1	24.9	1	1	100000	1	3	DS	2	1.17	1.50
Royal cucumber	5	3	2	7	1	1	24	2	1	35	2	1				BS	1	1.60	1.20
Sand sole	4	2	2	15	2	2	20	2	1	40	2	1	500000	1	2	BS	1	1.50	1.50
Greater forkbeard	3.5	2	2	20	3	2	22.5	2	1	110	2	2	684505	1	2	BS	1	2.00	1.67
European plaice	3	2	2	50	3	3	30.8	2	1	100	2	2	60000	2	3	BS	1	2.17	2.00
Pollack	4	2	2	15	2	2	41	3	1	130	3	2	323230	1	3	BS	1	2.00	1.83
Thornback ray	7	3	3	15	2	2	76.6	3	2	105	3	2	52	3	3	DS	2	2.67	2.33
Shagreen ray	8	3	3	12	2	2	85	3	2	120	3	2	45	3	3	DS	2	2.67	2.33
Spotted ray	5	3	2	18	2	2	61	3	2	83.5	3	2	24	3	3	DS	2	2.50	2.17
Undulate ray	8	3	3	23	3	2	81.2	3	2	100	3	2	45	3	3	DS	2	2.83	2.33
European pilchard	1	1	1	15	2	2	14.8	1	1	27.5	1	1	76000	2	3	BS	1	1.33	1.50
Atlantic chub mackerel	2	1	1	14	2	2	21.5	2	1	35.1	2	1	130000	1	3	BS	1	1.33	1.50
Mackerel	2	1	1	18	2	2	28.7	2	1	60	2	1	130000	1	3	BS	1	1.50	1.50
Catshark	4	2	2	12	2	2	52	3	2	100	3	2	105	3	3	DS	2	2.50	2.17
Elegant cuttlefish	1	1	1	1.5	1	1	4.2	1	1	25.2	1	1	513	3	3	DS	2	1.50	1.50
Cuttlefish	1	1	1	2	1	1	9	1	1	49	1	1	300	3	3	DS	2	1.67	1.50
Pink cuttlefish	0.7	1	1	0.8	1	1	7	1	1	42	1	1	201	3	3	DS	2	1.50	1.50
Sole	2.5	2	1	26	3	2	25	2	1	70	2	2	70000	2	3	BS	1	2.00	1.67
Spurdog	10.5	3	3	40	3	3	80	3	2	122	3	2	6	3	3	LB	3	3.00	2.67
European flying squid	1	1	1	2	1	1	23.2	2	1	75	2	2	205000	1	3	DS	2	1.50	1.67

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	Age at mat			Max age			Length at mat			Max length			Fecundity			Rep. strategy		Total score	
Species	Year	Q	K	Year	Q	K	Cm	Q	K	Cm	Q	K	egg/yr	Q	K	Code	Score	Q	K
Lesser flying squid	0.7	1	1	1	1	1	18.8	1	1	27	1	1	28000	2	3	DS	2	1.33	1.50
Greater weever	1	1	1	6	1	1	12	1	1	53	1	1	57600	2	3	BS	1	1.33	1.33
Mediterranean horse mackerel	2	1	1	10	1	1	20	2	1	60	2	1	77090	2	3	BS	1	1.50	1.33
Horse mackerel	3.5	2	2	20	3	2	23.9	2	1	70	2	2	77090	2	3	BS	1	2.00	1.83
Streaked gurnard	3.5	2	2	18	2	2	15	1	1	40	1	1	13000	2	3	BS	1	1.50	1.67
Pouting	2	1	1	4	1	1	21.6	2	1	46	2	1	207479	1	3	BS	1	1.33	1.33
Poor cod	1.2	1	1	5	1	1	13.4	1	1	40	1	1	1236	3	3	BS	1	1.33	1.33
John dory	5	3	2	12	2	2	36.2	3	1	90	3	2	30000	2	3	BS	1	2.33	1.83

C.3 Each gear susceptibility attributes values and scores

Table C.3: Bottom otter trawlers fleet susceptibility attributes values and scores (Sc), and presence in historical catches. Availability (global distribution, migration, and aggregation), encounterability (adult habitat, and bathymetry), selectivity (size at maturity, and Lc (length at 50% selectivity)), post capture mortality, total susceptability and historical catch presence.

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Greater argentine	2	1	3	3	HB,SB	3	3,4,5	3	3	38			3	3.00	No
Argentine	1	1	3	3	SB	3	3,4,5	3	3	13	13	1	3	2.28	Yes
Dragonet	1	2	2	2	SB	3	1,2,3	3	3	17	19	1	3	2.06	Yes
Brown crab	1	3	2	3	HB,SB	3	1,2,3	3	3	13			3	3.00	Yes
Boarfish	2	2	3	3	HB,SB	3	3,4,5	3	3	8	8	2	3	2.71	Yes
Red bandfish	1	2	2	2	SB	3	2,3,4	3	3	22	32	1	3	2.06	Yes
Red gurnard	2	1	3	3	HB,SB	3	3,4	3	3	27	20	2	3	2.71	Yes
Tub gurnard	1	3	3	3	HB,SB	3	2,3	3	3	22	20	2	3	2.71	Yes
Longfin gurnard	1	3	3	3	HB,SB	3	3,4	3	3	22	23	1	3	2.28	Yes
European conger	1	1	2	2	HB,SB	3	1,2,3,4,5	3	3	200	46	3	3	2.71	Yes
Sea bass	3	3	2	3	HB,SB	3	1,2,3	3	3	42	49	1	3	2.28	Yes
Wedge sole	1	2	2	2	SB	3	1,2,3	3	3	17	10	2	3	2.45	Yes
Common skate	3	2	2	3	SB	3	1,2,3	3	3	160			2	2.62	Yes
Longnose skate	2	2	2	2	SB	3	4,5	3	3	104			2	2.29	No
Curled octopus	1	2	2	2	HB,SB	3	1,2,3	3	3	11	30	1	3	2.06	Yes

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Table C.3 – Continued from previous page

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
European anchovy	3	3	2	3	EP	2	1,2,3,4	3	2	10	13	1	3	2.06	Yes
Grey gurnard	1	2	2	2	HB,SB	3	1,2,3	3	3	19	23	1	3	2.06	Yes
Tope shark	2	1	2	2	SB,EP	3	1,2,3,4	3	3	146	73	2	2	2.21	Yes
Blackbelly rosefish	1	2	2	2	SB	3	3,4,5	3	3	32	15	3	3	2.71	Yes
Broadtail shortfin	1	2	2	2	SB	3	3,4,5	3	3	21	17	2	3	2.45	Yes
Four-spot megrim	2	2	2	2	SB	3	3,4	3	3	20	18	2	3	2.45	Yes
Megrim	2	2	2	2	SB	3	3,4	3	3	20	14	2	3	2.45	Yes
Large-scaled gurnard	1	2	2	2	SB	3	3	3	3	8			3	2.62	Yes
Cuckoo ray	2	2	2	2	SB	3	2,3	3	3	58	39	2	2	2.21	Yes
Veined squid	1	2	2	2	HB,SB	3	3,4	3	3	40	9	3	3	2.71	Yes
European squid	1	2	2	2	HB,SB	3	2,3	3	3	28	21	2	3	2.45	Yes
Black anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	54	23	3	3	2.71	Yes
White anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	94	15	3	3	2.71	Yes
Haddock	3	1	3	3	SB	3	2,3	3	3	35	32	2	3	2.71	Yes
Whiting	3	2	3	3	HB,SB	3	2,3	3	3	28	16	2	3	2.71	Yes
Hake	2	1	3	3	SB	3	2,3,4	3	3	43	20	3	3	3.00	Yes
Thickback sole	1	2	3	3	SB	3	2,3,4	3	3	9	15	1	3	2.28	Yes
Blue whiting	2	1	3	3	EP	2	4	3	2	15	17	1	3	2.06	Yes

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Table C.3 – Continued from previous page

Species	Availability				Encounterability					Selectivity			Post mort	Total	
	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Ling	2	2	2	2	HB	1	3,4	3	1	90	48	2	3	1.86	Yes
Flathead grey mullet	1	3	3	3	SB	3	1	1	1	35	37	1	3	1.73	Yes
Red mullet	2	2	3	3	SB	3	1,2,3	3	3	22	15	2	3	2.71	Yes
Starry smooth-hound	2	3	2	3	SB	3	1,2,3	3	3	87	27	3	2	2.71	Yes
Common smooth-hound	2	1	2	2	SB	3	1,2,3	3	3	117	77	2	2	2.21	Yes
Norway lobster	3	2	2	3	SB	3	3,4,5	3	3	2			3	3.00	Yes
Common octopus	1	2	2	2	HB,SB	3	1,2,3	3	3	10			3	2.62	Yes
Royal cucumber	1	2	3	3	SB	3	3	3	3	24			3	3.00	Yes
Sand sole	1	2	2	2	SB	3	2,3	3	3	20	22	1	3	2.06	Yes
Greater forkbeard	2	2	3	3	SB	3	3,4	3	3	23	19	2	3	2.71	Yes
European plaice	3	3	2	3	SB	3	2,3	3	3	31	37	1	3	2.28	Yes
Pollack	3	1	3	3	HB	1	3	3	1	41	19	3	3	2.28	Yes
Thornback ray	3	3	2	3	HB,SB	3	2,3	3	3	77	29	3	2	2.71	Yes
Shagreen ray	3	3	2	3	HB,SB	3	3	3	3	85			2	2.62	Yes
Spotted ray	3	3	2	3	HB,SB	3	1,2,3	3	3	61	54	2	2	2.45	Yes
Undulate ray	3	3	2	3	HB,SB	3	1,2,3	3	3	81	56	2	2	2.45	Yes
European pilchard	3	1	3	3	EP	2	3	3	2	15	22	1	3	2.06	Yes
Atlantic chub mackerel	1	1	3	3	EP	2	1,2,3	3	2	22	24	1	3	2.06	Yes

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Table C.3 – Continued from previous page

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat		Bathymetry		Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
					code	Sc	code	Sc							
Mackerel	2	1	3	3	EP	2	1,2,3	3	2	29	31	1	3	2.06	Yes
Catshark	3	2	2	3	SB	3	3	3	3	52	41	2	2	2.45	Yes
Elegant cuttlefish	1	1	2	2	HB,SB	3	2,3	3	3	4	12	1	3	2.06	Yes
Cuttlefish	1	1	2	2	SB	3	1,2,3	3	3	9	20	1	3	2.06	Yes
Pink cuttlefish	1	1	2	2	SB	3	3	3	3	7	10	1	3	2.06	Yes
Sole	3	2	2	3	SB	3	2,3	3	3	25	12	3	3	3.00	Yes
Spurdog	2	1	2	2	EP,HB,SB	3	3	3	3	80	36	3	2	2.45	Yes
European flying squid	1	2	2	2	SB	3	1,2,3,4,5	3	3	23			3	2.62	Yes
Lesser flying squid	1	1	2	2	SB	3	2,3	3	3	19	14	2	3	2.45	Yes
Greater weever	1	2	2	2	SB	3	1,2	3	3	12	17	1	3	2.06	Yes
Mediterranean horse mackerel	1	1	3	3	EP	2	2,3	3	2	20	13	2	3	2.45	Yes
Horse mackerel	2	1	3	3	SB,EP	3	3,4	3	3	24	14	2	3	2.71	Yes
Streaked gurnard	1	2	2	2	HB,SB	3	1,2,3	3	3	15	25	1	3	2.06	Yes
Pouting	1	3	2	3	HB,SB	3	3	3	3	22	11	2	3	2.71	Yes
Poor cod	2	2	2	2	SB,EP	3	2,3	3	3	13	14	1	3	2.06	Yes
John dory	1	2	2	2	SB	3	3	3	3	36	12	3	3	2.71	Yes

Table C.4: Bottom pair trawlers fleet susceptibility attributes values and scores (Sc), and presence in historical catches. Availability (global distribution, migration, and aggregation), encounterability (adult habitat, and bathymetry), selectivity (size at maturity, and Lc (length at 50% selectivity)), post capture mortality, total susceptability and historical catch presence.

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Greater argentine	2	1	3	3	HB,SB	3	3,4,5	3	3	38	27	2	3	2.71	Yes
Argentine	1	1	3	3	SB	3	3,4,5	3	3	13	19	1	3	2.28	Yes
Dragonet	1	2	2	2	SB	3	1,2,3	3	3	17	27	1	3	2.06	Yes
Brown crab	1	3	2	3	HB,SB	3	1,2,3	3	3	13			3	3.00	Yes
Boarfish	2	2	3	3	HB,SB	3	3,4,5	3	3	8	11	1	3	2.28	Yes
Red bandfish	1	2	2	2	SB	3	2,3,4	3	3	22			3	2.62	Yes
Red gurnard	2	1	3	3	HB,SB	3	3,4	3	3	27	22	2	3	2.71	Yes
Tub gurnard	1	3	3	3	HB,SB	3	2,3	3	3	22	26	1	3	2.28	Yes
Longfin gurnard	1	3	3	3	HB,SB	3	3,4	3	3	22	27	1	3	2.28	Yes
European conger	1	1	2	2	HB,SB	3	1,2,3,4,5	3	3	200	52	3	3	2.71	Yes
Sea bass	3	3	2	3	HB,SB	3	1,2,3	3	3	42	57	1	3	2.28	Yes
Wedge sole	1	2	2	2	SB	3	1,2,3	3	3	17			3	2.62	No
Common skate	3	2	2	3	SB	3	1,2,3	3	3	160			2	2.62	No
Longnose skate	2	2	2	2	SB	3	4,5	3	3	104			2	2.29	No
Curled octopus	1	2	2	2	HB,SB	3	1,2,3	3	3	11			3	2.62	Yes
European anchovy	3	3	2	3	EP	2	1,2,3,4	3	2	10	12	1	3	2.06	Yes
Grey gurnard	1	2	2	2	HB,SB	3	1,2,3	3	3	19	23	1	3	2.06	Yes

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Table C.4 – Continued from previous page

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Tope shark	2	1	2	2	SB,EP	3	1,2,3,4	3	3	146	121	2	2	2.21	Yes
Blackbelly rosefish	1	2	2	2	SB	3	3,4,5	3	3	32	26	2	3	2.45	Yes
Broadtail shortfin	1	2	2	2	SB	3	3,4,5	3	3	21	16	2	3	2.45	Yes
Four-spot megrim	2	2	2	2	SB	3	3,4	3	3	20	29	1	3	2.06	Yes
Megrim	2	2	2	2	SB	3	3,4	3	3	20	27	1	3	2.06	Yes
Large-scaled gurnard	1	2	2	2	SB	3	3	3	3	8			3	2.62	No
Cuckoo ray	2	2	2	2	SB	3	2,3	3	3	58	54	2	2	2.21	Yes
Veined squid	1	2	2	2	HB,SB	3	3,4	3	3	40	20	2	3	2.45	Yes
European squid	1	2	2	2	HB,SB	3	2,3	3	3	28	24	2	3	2.45	Yes
Black anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	54	16	3	3	2.71	Yes
White anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	94	52	2	3	2.45	Yes
Haddock	3	1	3	3	SB	3	2,3	3	3	35	38	1	3	2.28	Yes
Whiting	3	2	3	3	HB,SB	3	2,3	3	3	28	35	1	3	2.28	Yes
Hake	2	1	3	3	SB	3	2,3,4	3	3	43	24	2	3	2.71	Yes
Thickback sole	1	2	3	3	SB	3	2,3,4	3	3	9	20	1	3	2.28	No
Blue whiting	2	1	3	3	EP	2	4	3	2	15	22	1	3	2.06	Yes
Ling	2	2	2	2	HB	1	3,4	3	1	90	76	2	3	1.86	Yes
Flathead grey mullet	1	3	3	3	SB	3	1	1	1	35			3	2.08	No

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Table C.4 – Continued from previous page

Species	Availability				Encounterability					Selectivity			Post mort	Total	
	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Red mullet	2	2	3	3	SB	3	1,2,3	3	3	22	18	2	3	2.71	Yes
Starry smooth-hound	2	3	2	3	SB	3	1,2,3	3	3	87	68	2	2	2.45	Yes
Common smooth-hound	2	1	2	2	SB	3	1,2,3	3	3	117	77	2	2	2.21	Yes
Norway lobster	3	2	2	3	SB	3	3,4,5	3	3	2			3	3.00	No
Common octopus	1	2	2	2	HB,SB	3	1,2,3	3	3	10			3	2.62	No
Royal cucumber	1	2	3	3	SB	3	3	3	3	24			3	3.00	No
Sand sole	1	2	2	2	SB	3	2,3	3	3	20			3	2.62	No
Greater forkbeard	2	2	3	3	SB	3	3,4	3	3	23	25	1	3	2.28	Yes
European plaice	3	3	2	3	SB	3	2,3	3	3	31			3	3.00	No
Pollack	3	1	3	3	HB	1	3	3	1	41	72	1	3	1.73	Yes
Thornback ray	3	3	2	3	HB,SB	3	2,3	3	3	77	25	3	2	2.71	Yes
Shagreen ray	3	3	2	3	HB,SB	3	3	3	3	85	83	2	2	2.45	Yes
Spotted ray	3	3	2	3	HB,SB	3	1,2,3	3	3	61	102	1	2	2.06	No
Undulate ray	3	3	2	3	HB,SB	3	1,2,3	3	3	81			2	2.62	No
European pilchard	3	1	3	3	EP	2	3	3	2	15	22	1	3	2.06	Yes
Atlantic chub mackerel	1	1	3	3	EP	2	1,2,3	3	2	22	30	1	3	2.06	Yes
Mackerel	2	1	3	3	EP	2	1,2,3	3	2	29	19	2	3	2.45	Yes
Catshark	3	2	2	3	SB	3	3	3	3	52	46	2	2	2.45	Yes

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Table C.4 – Continued from previous page

	Availability				Encounterability					Selectivity			Post mort	Total	
Species	Global dist score	Mig score	Agg score	Sc	Adult habitat code	Adult habitat Sc	Bathymetry code	Bathymetry Sc	Sc	Size at mat (cm)	Lc (cm)	Sc	Sc	Sc	Hist catch
Elegant cuttlefish	1	1	2	2	HB,SB	3	2,3	3	3	4			3	2.62	Yes
Cuttlefish	1	1	2	2	SB	3	1,2,3	3	3	9	12	1	3	2.06	Yes
Pink cuttlefish	1	1	2	2	SB	3	3	3	3	7	9	1	3	2.06	Yes
Sole	3	2	2	3	SB	3	2,3	3	3	25	28	1	3	2.28	Yes
Spurdog	2	1	2	2	EP,HB,SB	3	3	3	3	80	58	2	2	2.21	Yes
European flying squid	1	2	2	2	SB	3	1,2,3,4,5	3	3	23			3	2.62	Yes
Lesser flying squid	1	1	2	2	SB	3	2,3	3	3	19	11	2	3	2.45	Yes
Greater weever	1	2	2	2	SB	3	1,2	1	1	12	23	1	3	1.57	Yes
Mediterranean horse mackerel	1	1	3	3	EP	2	2,3	3	2	20	22	1	3	2.06	Yes
Horse mackerel	2	1	3	3	SB,EP	3	3,4	3	3	24	22	2	3	2.71	Yes
Streaked gurnard	1	2	2	2	HB,SB	3	1,2,3	3	3	15	24	1	3	2.06	Yes
Pouting	1	3	2	3	HB,SB	3	3	3	3	22	16	2	3	2.71	Yes
Poor cod	2	2	2	2	SB,EP	3	2,3	3	3	13	15	1	3	2.06	Yes
John dory	1	2	2	2	SB	3	3	3	3	36	11	3	3	2.71	Yes

Table C.5: Longliners fleet susceptibility attributes values and scores (Sc), and presence in historical catches. Availability (global distribution, migration, and aggregation), encounterability (adult habitat, and bathymetry), selectivity (size at maturity, and Lc (length at 50% selectivity)), post capture mortality, total susceptability and historical catch presence.

	Availability				Encounterability					Post mort	Total	
					Adult habitat		Bathymetry					
Species	Global dist score	Mig score	Agg score	Sc	code	Sc	code	Sc	Sc	Sc	Sc	Hist catch
Greater argentine	2	1	3	3	HB,SB	3	3,4,5	3	3	3	3.00	No
Argentine	1	1	3	3	SB	3	3,4,5	3	3	3	3.00	No
Dragonet	1	2	2	2	SB	3	1,2,3	1	1	3	1.82	No
Brown crab	1	3	2	3	HB,SB	3	1,2,3	1	1	3	2.08	No
Boarfish	2	2	3	3	HB,SB	3	3,4,5	3	3	3	3.00	No
Red bandfish	1	2	2	2	SB	3	2,3,4	3	3	3	2.62	Yes
Red gurnard	2	1	3	3	HB,SB	3	3,4	3	3	3	3.00	No
Tub gurnard	1	3	3	3	HB,SB	3	2,3	1	1	3	2.08	No
Longfin gurnard	1	3	3	3	HB,SB	3	3,4	3	3	3	3.00	No
European conger	1	1	2	2	HB,SB	3	1,2,3,4,5	3	3	3	2.62	Yes
Sea bass	3	3	2	3	HB,SB	3	1,2,3	1	1	3	2.08	Yes
Wedge sole	1	2	2	2	SB	3	1,2,3	1	1	3	1.82	No
Common skate	3	2	2	3	SB	3	1,2,3	1	1	2	1.82	No
Longnose skate	2	2	2	2	SB	3	4,5	3	3	2	2.29	Yes
Curled octopus	1	2	2	2	HB,SB	3	1,2,3	1	1	3	1.82	No
European anchovy	3	3	2	3	EP	2	1,2,3,4	3	2	3	2.62	No
Grey gurnard	1	2	2	2	HB,SB	3	1,2,3	1	1	3	1.82	No
Tope shark	2	1	2	2	SB,EP	3	1,2,3,4	3	3	2	2.29	Yes

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Table C.5 – Continued from previous page

	Availability				Encounterability					Post mort	Total	
	Global dist	Mig	Agg	Sc	Adult habitat		Bathymetry		Sc	Sc	Sc	Hist catch
Species	score	score	score		code	Sc	code	Sc				
Blackbelly rosefish	1	2	2	2	SB	3	3,4,5	3	3	3	2.62	Yes
Broadtail shortfin	1	2	2	2	SB	3	3,4,5	3	3	3	2.62	No
Four-spot megrim	2	2	2	2	SB	3	3,4	3	3	3	2.62	No
Megrim	2	2	2	2	SB	3	3,4	3	3	3	2.62	No
Large-scaled gurnard	1	2	2	2	SB	3	3	1	1	3	1.82	No
Cuckoo ray	2	2	2	2	SB	3	2,3	1	1	2	1.59	Yes
Veined squid	1	2	2	2	HB,SB	3	3,4	3	3	3	2.62	No
European squid	1	2	2	2	HB,SB	3	2,3	1	1	3	1.82	No
Black anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	3	2.62	Yes
White anglerfish	2	2	2	2	HB,SB	3	3,4,5	3	3	3	2.62	Yes
Haddock	3	1	3	3	SB	3	2,3	1	1	3	2.08	Yes
Whiting	3	2	3	3	HB,SB	3	2,3	1	1	3	2.08	Yes
Hake	2	1	3	3	SB	3	2,3,4	3	3	3	3.00	Yes
Thickback sole	1	2	3	3	SB	3	2,3,4	3	3	3	3.00	No
Blue whiting	2	1	3	3	EP	2	4	3	2	3	2.62	No
Ling	2	2	2	2	HB	2	3,4	3	2	3	2.29	Yes
Flathead grey mullet	1	3	3	3	SB	3	1	1	1	3	2.08	No
Red mullet	2	2	3	3	SB	3	1,2,3	1	1	3	2.08	No
Starry smooth-hound	2	3	2	3	SB	3	1,2,3	1	1	2	1.82	No

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Table C.5 – Continued from previous page

	Availability				Encounterability					Post mort	Total	
	Global dist	Mig	Agg		Adult habitat		Bathymetry					
Species	score	score	score	Sc	code	Sc	code	Sc	Sc	Sc	Sc	Hist catch
Common smooth-hound	2	1	2	2	SB	3	1,2,3	1	1	2	1.59	Yes
Norway lobster	3	2	2	3	SB	3	3,4,5	3	3	3	3.00	No
Common octopus	1	2	2	2	HB,SB	3	1,2,3	1	1	3	1.82	No
Royal cucumber	1	2	3	3	SB	3	3	1	1	3	2.08	No
Sand sole	1	2	2	2	SB	3	2,3	1	1	3	1.82	No
Greater forkbeard	2	2	3	3	SB	3	3,4	3	3	3	3.00	Yes
European plaice	3	3	2	3	SB	3	2,3	1	1	3	2.08	No
Pollack	3	1	3	3	HB	2	3	1	1	3	2.08	Yes
Thornback ray	3	3	2	3	HB,SB	3	2,3	1	1	2	1.82	Yes
Shagreen ray	3	3	2	3	HB,SB	3	3	1	1	2	1.82	Yes
Spotted ray	3	3	2	3	HB,SB	3	1,2,3	1	1	2	1.82	Yes
Undulate ray	3	3	2	3	HB,SB	3	1,2,3	1	1	2	1.82	Yes
European pilchard	3	1	3	3	EP	2	3	1	1	3	2.08	No
Atlantic chub mackerel	1	1	3	3	EP	2	1,2,3	1	1	3	2.08	No
Mackerel	2	1	3	3	EP	2	1,2,3	1	1	3	2.08	No
Catshark	3	2	2	3	SB	3	3	1	1	2	1.82	Yes
Elegant cuttlefish	1	1	2	2	HB,SB	3	2,3	1	1	3	1.82	No
Cuttlefish	1	1	2	2	SB	3	1,2,3	1	1	3	1.82	No
Pink cuttlefish	1	1	2	2	SB	3	3	1	1	3	1.82	No

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Table C.5 – Continued from previous page

	Availability				Encounterability					Post mort	Total	
	Global dist	Mig	Agg		Adult habitat		Bathymetry					
Species	score	score	score	Sc	code	Sc	code	Sc	Sc	Sc	Sc	Hist catch
Sole	3	2	2	3	SB	3	2,3	1	1	3	2.08	No
Spurdog	2	1	2	2	EP,HB,SB	3	3	1	1	2	1.59	Yes
European flying squid	1	2	2	2	SB	3	1,2,3,4,5	3	3	3	2.62	No
Lesser flying squid	1	1	2	2	SB	3	2,3	1	1	3	1.82	No
Greater weever	1	2	2	2	SB	3	1,2	1	1	3	1.82	No
Mediterranean horse mackerel	1	1	3	3	EP	2	2,3	1	1	3	2.08	No
Horse mackerel	2	1	3	3	SB,EP	3	3,4	3	3	3	3.00	No
Streaked gurnard	1	2	2	2	HB,SB	3	1,2,3	1	1	3	1.82	No
Pouting	1	3	2	3	HB,SB	3	3	1	1	3	2.08	Yes
Poor cod	2	2	2	2	SB,EP	3	2,3	1	1	3	1.82	No
John dory	1	2	2	2	SB	3	3	1	1	3	1.82	Yes

C.4 Data quality score and source of each productivity and susceptibility attributes

Table C.6: Data quality score and source of each productivity attribute. S: Score, R: Information source used; BR: British sea fishing (2019), FB: FishBase (2019), IC: ICES latest report (2019), IU: IUCN Red list (2019), SL: SeaLifeBase (2019), AC: Akyol and Çoker (2001), AF: Abdallah and Faltas (1988), AG: Ak and Genç (2013), AM: Amin et al. (2016), BE: Beukhof et al. (2019), CO: Choen et al. (1990), DU: Dursun et al. (2013), FE: Felix et al. (2011), FR: Fernandez-Arcaya et al. (2013), GR: Guerra and Roncha (1994), HA: Hastie et al. (2009), HE: Herrera et al. (2008), IL: İlkyaz et al. (2010), IS: Ismen et al. (2004), KA: Kaya et al. (2001), LA: Laptikhovsky (2000), LO: Lordan et al. (2001), MA: MarLIN (2006), MO: Mori et al. (2001), MU: Mueller (2016), OR: Orsi Relini et al. (2006), OS: O'Sullivan et al. (2003), SA: Salman (2015), SA2: Salman (2017), UN: Ungfors (2007), VD: Vidal et al. (2014), VI: Viette et al. (1997), WH: Walker and Hislop (1998), EX: Expert opinion, ND: No data.

	Productivity attributes											
Species	Maximum size		Age at maturity		Size at maturity		Maximum age		Reproductive strategy		Fecundity	
	S	R	S	R	S	R	S	R	S	R	S	R
Greater argentine	2	IU	2	IU	2	IU	2	IU	1	FB	2	FB
Argentine	2	IU	2	BE	2	IU	2	IU	1	FB	3	FB
Dragonet	2	FB	2	BE	3	FB	2	IU	1	FB	3	MA
Brown crab	2	SL	3	MA	2	UN	2	MA	1	FB	3	MA
Boarfish	2	IU	2	IU	2	IU	2	IU	1	FB	5	ND
Red bandfish	2	IU	3	KA	3	IU	3	KA	1	IU	5	ND
Red gurnard	3	FB	3	BE	3	FB	3	FB	1	FB	3	IS
Tub gurnard	3	IU	3	IU	3	IU	3	IU	1	FB	2	IS
Longfin gurnard	2	FB	4	EX	4	EX	4	EX	3	FB	3	IS
European conger	3	IU	2	BR	2	FB	3	OS	1	BR	2	FB
Sea bass	1	IC	2	IC	1	IC	2	IC	1	FB	3	FB
Wedge sole	2	FB	4	FB	2	FB	4	FB	3	IU	2	HE
Common skate	2	FB	3	BE	2	FB	3	BE	1	IU	2	MA +FB

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Table C.6 – Continued from previous page

Species	Productivity attributes											
	Maximum size		Age at maturity		Size at maturity		Maximum age		Reproductive strategy		Fecundity	
	S	R	S	R	S	R	S	R	S	R	S	R
Longnose skate	2	IU	4	EX	2	IU	2	IU	1	FB	3	MA +FB
Curled octopus	2	SL	2	OR	2	SL	2	OR	1	SL	5	ND
European anchovy	2	FB	1	IC	2	FB	2	FB	1	IC	1	FB
Grey gurnard	2	IU	2	FB	2	FB	2	IU	1	FB	2	FB
Tope shark	3	FB	3	BE	3	FB	3	FB	1	FB	3	FB
Blackbelly rosefish	3	FB	2	FB	3	FB	3	FB	1	FB	2	FB
Broadtail shortfin	2	SL	4	EX	2	SL	4	EX	1	SL	3	SA2
Four-spot megrim	3	FB	3	IC	3	IC	3	IC	1	FB	2	FB
Megrim	1	IC	1	IC	1	IC	1	IC	1	FB	2	FB
Large-scaled gurnard	3	FB	3	FB	3	FB	4	EX	3	FB	3	IL
Cuckoo ray	2	FB	2	FB	2	FB	3	BE	1	FB	2	FB
Veined squid	3	GR	3	SL	2	GR	2	GR	1	SL	3	LA
European squid	3	GR	3	SL	2	GR	2	GR	1	SL	3	LA
Black anglerfish	1	IC	3	BE	2	IC	2	FB	1	FB	3	FB
White anglerfish	2	FB	2	IU	2	IC	1	IC	1	FB	2	FB
Haddock	3	FB	3	IU	3	FB	3	FB	1	FB	2	FB
Whiting	2	FB	2	FB	2	FB	2	FB	1	FB	2	FB
Hake	2	FB	2	FB	1	IC	2	FB	1	FB	1	FB
Thickback sole	2	IU	2	BE	2	BE	2	IU	3	FB	2	FE

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Table C.6 – Continued from previous page

Species	Productivity attributes											
	Maximum size		Age at maturity		Size at maturity		Maximum age		Reproductive strategy		Fecundity	
	S	R	S	R	S	R	S	R	S	R	S	R
Blue whiting	2	IU	2	IU	2	FB	2	IU	1	FB	2	FB
Ling	2	FB	2	IU	2	FB	2	FB	1	FB	5	ND
Flathead grey mullet	2	FB	3	FB	2	FB	2	FB	1	FB	5	ND
Red mullet	1	IC	2	IC	1	IC	1	IC	1	FB	3	AM
Starry smooth-hound	1	IC	2	IC	2	IC	2	IC	1	FB	3	FB
Common smooth-hound	3	IC	2	IU	3	IC	3	IC	1	FB	3	FB
Norway lobster	2	IU	2	MA	1	IC	2	MA	1	SL	3	MO
Common octopus	2	SL	4	EX	4	EX	4	EX	1	SL	3	VD
Royal cucumber	2	SL	4	EX	4	EX	3	MU	1	SL	5	ND
Sand sole	2	FB	4	EX	4	EX	2	FB	3	FB	2	FE
Greater forkbeard	2	FB	3	BE	3	BE	2	FB	3	FB	2	FR
European plaice	2	FB	2	FB	2	FB	2	FB	1	FB	3	MA
Pollack	2	FB	2	IU	2	FB	2	IU	1	FB	3	CO
Thornback ray	2	FB	3	FB	2	FB	2	FB	1	FB	3	FB
Shagreen ray	2	FB	4	EX	2	WH	4	EX	1	FB	4	FB
Spotted ray	2	FB	3	BE	2	FB	2	FB	1	FB	3	FB
Undulate ray	2	FB	4	EX	1	IC	2	IU	1	FB	4	FB
European pilchard	2	FB	2	FB	2	FB	2	FB	1	FB	1	FB
Atlantic chub mackerel	2	FB	4	EX	2	FB	2	IU	1	FB	3	FB

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Table C.6 – Continued from previous page

Species	Productivity attributes											
	Maximum size		Age at maturity		Size at maturity		Maximum age		Reproductive strategy		Fecundity	
	S	R	S	R	S	R	S	R	S	R	S	R
Mackerel	2	FB	1	IC	2	FB	2	IU	1	FB	2	FB
Catshark	2	FB	2	FB	1	IC	2	FB	1	FB	2	FB
Elegant cuttlefish	3	SL	2	SL	2	SA	2	SL	1	SL	3	SA
Cuttlefish	2	SL	2	SL	2	SL	2	SL	1	SL	3	SA
Pink cuttlefish	2	SL	4	EX	2	DU	2	SL	1	SL	3	DU
Sole	2	FB	1	IC	1	IC	2	FB	1	FB	1	FB
Spurdog	2	FB	3	BE	1	IC	2	IU	1	FB	2	FB
European flying squid	2	SL	4	EX	2	SL	2	SL	1	SL	2	LO
Lesser flying squid	2	SL	4	EX	2	SL	4	SL	1	SL	2	HA
Greater weever	2	IU	2	BE	2	BE	2	IU	1	FB	3	AG
Mediterranean horse mackerel	2	FB	2	VI	2	FB	4	IC	1	FB	3	FB
Horse mackerel	2	FB	1	IC	2	FB	1	IC	1	FB	1	FB
Streaked gurnard	2	FB	2	IU	2	FB	2	FB	1	FB	3	AF
Pouting	2	FB	2	BE	2	FB	2	FB	1	FB	2	FB
Poor cod	2	FB	3	FB	2	FB	2	FB	1	FB	3	FB
John dory	2	FB	3	IU	2	FB	2	FB	1	FB	3	AC

Literature:
BR: `www.britishseafishing.co.uk`
FB: `www.fishbase.org`
IC: `www.ices.dk`

IU: `www.iucnredlist.org`
SL: `www.sealifebase.org`
AC: Akyol O., and Çoker T. (2001) A preliminary study on determination of batch fecundity of John Dory. Zeus faber. 1:161–172
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LA: Laptikhovsky, V. 2000. Fecundity of the squid Loligo vulgaris Lamarck, 1798 (Myopsida, Loliginidae) off Northwest Africa. Scientia Marina, 64: 275-278.
LO: Lordan, C., Collins, M. A., Key, L. N., and Browne, E. 2001. Biology of the squid Todarodessagittatus for Irish and Scottish waters. Journal of the Marine Biological Association of the United Kingdom, 81: 299-306.
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Table C.7: Data quality score and source of each susceptibility attribute. Length at 50% selectivity (Lc) by gear; bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). S: score, R: information source used; FB: FishBase (2019), IC: ICES latest report (2019), IU: IUCN Red list (2019), SL: SeaLifeBase (2019), AZ: AZTI fisheries database, MA: MarLIN (2006), EX: expert opinion, ND: no data.

	Susceptibility attributes																	
Species	Global Distribution		Migration		Aggegation		Adult habitat		Bathymetry		Post capture mortality		Lc OTB		Lc PTB		Lc LLS	
	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R
Greater argentine	1	IC	2	IU	2	IU	2	IU	2	IU	4	EX	5	ND	2	AZ	5	ND
Argentine	3	IU	3	IU	3	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Dragonet	3	IU	2	MA	2	MA	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Brown crab	3	SL	2	SL	2	SL	2	SL	2	SL	4	EX	5	ND	5	ND	5	ND
Boarfish	1	IC	4	EX	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Red bandfish	3	IU	4	EX	3	IU	2	IU	2	IU	4	EX	2	AZ	5	ND	5	ND
Red gurnard	1	IC	1	IC	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Tub gurnard	3	IU	3	IU	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Longfin gurnard	3	IU	4	EX	3	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
European conger	3	IU	2	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Sea bass	1	IC	2	IU	2	IU	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Wedge sole	3	IU	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	5	ND	5	ND
Common skate	1	IC	4	EX	4	EX	3	IU	2	IU	4	EX	5	ND	5	ND	5	ND
Longnose skate	3	IU	4	EX	4	EX	2	IU	2	IU	4	EX	5	ND	5	ND	5	ND
Curled octopus	3	IU	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	5	ND	5	ND
European anchovy	1	IC	2	IU	1	IC	1	IC	2	IU	4	EX	2	AZ	2	AZ	5	ND
Grey gurnard	3	IU	4	EX	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Tope shark	1	IC	2	IU	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND

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Table C.7 – Continued from previous page

Species	Susceptibility attributes																	
	Global Distribution		Migration		Aggegation		Adult habitat		Bathymetry		Post capture mortality		Lc OTB		Lc PTB		Lc LLS	
	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R
Blackbelly rosefish	3	FB	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Broadtail shortfin	3	IU	4	EX	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Four-spot megrim	1	IC	1	IC	4	EX	2	IU	1	IC	4	EX	2	AZ	2	AZ	5	ND
Megrim	1	IC	1	IC	4	EX	2	IU	1	IC	4	EX	2	AZ	2	AZ	5	ND
Large-scaled gurnard	3	FB	4	EX	4	EX	2	IU	2	IU	4	EX	5	ND	5	ND	5	ND
Cuckoo ray	1	IC	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Veined squid	3	IU	4	EX	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
European squid	3	IU	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Black anglerfish	1	IC	4	EX	4	EX	3	IU	1	IC	4	EX	2	AZ	2	AZ	5	ND
White anglerfish	1	IC	4	EX	4	EX	2	IU	1	IC	4	EX	2	AZ	2	AZ	5	ND
Haddock	3	IC	2	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Whiting	1	IC	4	EX	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Hake	1	IC	1	IC	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Thickback sole	3	IU	4	EX	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Blue whiting	1	IC	1	IC	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Ling	1	IC	4	EX	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Flathead grey mullet	3	FB	2	FB	2	FB	2	FB	2	FB	4	EX	2	AZ	5	ND	5	ND
Red mullet	1	IC	4	EX	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Starry smooth-hound	1	IC	1	IC + IU	4	EX	2	FB	2	IU	4	EX	2	AZ	2	AZ	5	ND

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Table C.7 – Continued from previous page

Species	Global Distribution		Migration		Aggegation		Adult habitat		Bathymetry		Post capture mortality		Lc OTB		Lc PTB		Lc LLS	
	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R
Common smooth-hound	1	IC	3	IC	4	EX	2	FB	2	IU	4	EX	2	AZ	2	AZ	5	ND
Norway lobster	1	IC	2	MA	2	MA	2	IU	2	FB	4	EX	5	ND	5	ND	5	ND
Common octopus	3	SL	2	IU	4	EX	2	SL	2	SL	4	EX	5	ND	5	ND	5	ND
Royal cucumber	3	SL	4	EX	4	EX	2	SL	2	SL	4	EX	5	ND	5	ND	5	ND
Sand sole	3	FB	4	EX	4	EX	2	FB	2	FB	4	EX	2	AZ	5	ND	5	ND
Greater forkbeard	2	IC	2	FB	4	EX	2	FB	2	FB	4	EX	2	AZ	2	AZ	5	ND
European plaice	1	IC	2	FB	4	EX	2	FB	2	FB	4	EX	2	AZ	5	ND	5	ND
Pollack	1	IC	2	FB	4	EX	2	FB	2	FB	4	EX	2	AZ	2	AZ	5	ND
Thornback ray	1	IC	2	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Shagreen ray	3	IC	3	IU	4	EX	2	IU	2	FB	4	EX	5	ND	2	AZ	5	ND
Spotted ray	1	IC	3	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Undulate ray	1	IC	2	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	5	ND	5	ND
European pilchard	1	IC	2	IU	2	FB	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Atlantic chub mackerel	3	IU	3	IU	2	IU	3	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Mackerel	1	IC	2	IU	2	IU	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Catshark	1	IC	4	EX	2	IU	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Elegant cuttlefish	3	IU	3	IU	4	EX	2	IU	2	SL	4	EX	2	AZ	5	ND	5	ND
Cuttlefish	3	SL	2	IU	4	EX	2	IU	2	SL	4	EX	2	AZ	2	AZ	5	ND
Pink cuttlefish	3	IU	3	IC	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND

The column group "Global Distribution" through "Lc LLS" is headed "Susceptibility attributes".

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Table C.7 – Continued from previous page

Species	Susceptibility attributes																	
	Global Distribution		Migration		Aggegation		Adult habitat		Bathymetry		Post capture mortality		Lc OTB		Lc PTB		Lc LLS	
	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R	S	R
Sole	1	IC	1	IC	4	EX	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
Spurdog	1	IC	1	IU	2	IU	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
European flying squid	3	IU	2	IU	2	IU	2	IU	2	IU	4	EX	5	ND	5	ND	5	ND
Lesser flying squid	3	SL	3	IU	4	EX	2	IU	2	IU	4	EX	2	AZ	2	AZ	5	ND
Greater weever	3	IU	4	EX	4	EX	2	FB	2	IU	4	EX	2	AZ	2	AZ	5	ND
Mediterranean horse mackerel	3	IU	2	FB	2	FB	2	FB	2	IU	4	EX	2	AZ	2	AZ	5	ND
Horse mackerel	1	IC	1	IC	2	FB	2	FB	2	FB	4	EX	2	AZ	2	AZ	5	ND
Streaked gurnard	3	IU	4	EX	4	EX	2	FB	2	FB	4	EX	2	AZ	2	AZ	5	ND
Pouting	3	IU	2	IU	2	IU	3	FB	2	FB	4	EX	2	AZ	2	AZ	5	ND
Poor cod	3	FB	2	FB	2	FB	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND
John dory	3	IU	4	EX	2	FB	2	IU	2	FB	4	EX	2	AZ	2	AZ	5	ND

Literature*:
FB: www.fishbase.org
IC: www.ices.dk
IU: www.iucnredlist.org
SL: www.sealifebase.org
MA: MarLIN, 2006. BIOTIC - Biological Traits Information Catalogue. Marine Life Information Network. Plymouth: Marine Biological Association of the United Kingdom. Downloaded on 16 August 2019. Available from www.marlin.ac.uk/biotic.

C.5 Potential risk scores and their corresponding data quality index

Table C.8: Potential risk score and their corresponding data quality score for bottom otter trawlers (OTB), bottom pair trawlers (PTB), longliners (LLS) and overall fishery (AS) fleets. Data risk scores; low <2.64, medium 2.64-3.18, high >3.18. Data quality scores: poor >3.5, moderate 2.0-3.5, and good <2.0. Empty means that a fleet had no catches associated to a species.

	Risk Score				Data quality Score			
Species	OTB	PTB	LLS	AS	OTB	PTB	LLS	AS
Greater argentine		3.69		3.69		2.13		2.13
Argentine	2.82	2.82		3.27	2.38	2.38		2.38
Dragonet	2.45	2.45		2.83	2.44	2.44		2.44
Brown crab	3.52	3.52		3.52	2.56	2.56		2.56
Boarfish	3.25	2.90		3.50	2.46	2.46		2.46
Red bandfish	2.74	3.18	3.18	3.50	2.90	3.08	3.08	3.02
Red gurnard	3.27	3.27		3.52	2.56	2.56		2.56
Tub gurnard	3.18	2.82		3.43	2.65	2.65		2.65
Longfin gurnard	2.82	2.82		3.27	3.21	3.21		3.21
European conger	3.58	3.58	3.51	3.80	2.46	2.46	2.65	2.52
Sea bass	3.26	3.26	3.13	3.80	1.98	1.98	2.17	2.04
Wedge sole	2.96			2.96	2.88			2.88
Common skate	3.86			3.86	2.71			2.71
Longnose skate			3.39	3.39			2.81	2.81
Curled octopus	2.49	2.97		3.25	2.63	2.81		2.72
European anchovy	2.29	2.29		2.69	1.85	1.85		1.85
Grey gurnard	2.37	2.37		2.76	2.29	2.29		2.29
Tope shark	3.73	3.73	3.77	4.24	2.69	2.69	2.88	2.75
Blackbelly rosefish	3.58	3.38	3.51	3.80	2.69	2.69	2.88	2.75
Broadtail shortfin	2.87	2.87		3.35	2.71	2.71		2.71
Four-spot megrim	2.96	2.65		3.25	2.50	2.50		2.50
Megrim	3.06	2.76		3.34	1.71	1.71		1.71
Large-scaled gurnard	2.94			2.94	3.29			3.29
Cuckoo ray	3.34	3.34	2.96	3.76	2.38	2.38	2.56	2.44
Veined squid	3.37	3.16		3.61	2.63	2.63		2.63
European squid	3.06	3.06		3.52	2.63	2.63		2.63
Black anglerfish	3.69	3.69	3.62	3.91	2.38	2.38	2.56	2.44
White anglerfish	3.58	3.38	3.51	3.80	2.15	2.15	2.33	2.21
Haddock	3.47	3.14	3.00	3.70	2.69	2.69	2.88	2.75
Whiting	3.27	2.93	2.77	3.52	2.29	2.29	2.48	2.35
Hake	3.80	3.58	3.80	3.80	1.94	1.94	2.13	2.00
Thickback sole	2.64			2.64	2.54			2.54
Blue whiting	2.76	2.76		3.10	2.17	2.17		2.17
Ling	3.04	3.04	3.32	3.67	2.54	2.54	2.73	2.60

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Table C.8 – Continued from previous page

	Risk Score				Data quality Score			
Species	OTB	PTB	LLS	AS	OTB	PTB	LLS	AS
Flathead grey mullet	2.80			2.80	2.54			2.54
Red mullet	3.10	3.10		3.35	2.06	2.06		2.06
Starry smooth-hound	3.92	3.75		4.13	2.17	2.17		2.17
Common smooth-hound	3.73	3.73	3.39	4.11	2.65	2.65	2.83	2.71
Norway lobster	3.43			3.43	2.25			2.25
Common octopus	2.87			2.87	3.19			3.19
Royal cucumber	3.40			3.40	3.35			3.35
Sand sole	2.55			2.55	3.00			3.00
Greater forkbeard	3.37	3.03	3.61	3.61	2.65	2.65	2.83	2.71
European plaice	3.14			3.14	2.29			2.29
Pollack	3.03	2.65	2.89	3.46	2.29	2.29	2.48	2.35
Thornback ray	3.80	3.80	3.23	4.01	2.38	2.38	2.56	2.44
Shagreen ray	3.74	3.62	3.23	4.01	3.02	2.83	3.02	2.96
Spotted ray	3.50		3.09	3.65	2.42		2.60	2.51
Undulate ray	3.75		3.37	3.89	2.40		2.58	2.49
European pilchard	2.45	2.45		2.83	2.04	2.04		2.04
Atlantic chub mackerel	2.45	2.45		2.83	2.56	2.56		2.56
Mackerel	2.55	2.87		3.17	2.04	2.04		2.04
Catshark	3.50	3.50	3.09	3.91	2.06	2.06	2.25	2.13
Elegant cuttlefish	2.55	3.02		3.30	2.50	2.69		2.59
Cuttlefish	2.65	2.65		3.00	2.38	2.38		2.38
Pink cuttlefish	2.55	2.55		2.91	2.58	2.58		2.58
Sole	3.61	3.03		3.61	1.85	1.85		1.85
Spurdog	3.87	3.73	3.39	4.23	2.02	2.02	2.21	2.08
European flying squid	3.02	3.02		3.35	2.56	2.56		2.56
Lesser flying squid	2.79	2.79		3.28	2.67	2.67		2.67
Greater weever	2.45	2.06		2.57	2.46	2.46		2.46
Mediterranean horse mackerel	2.87	2.55		3.17	2.46	2.46		2.46
Horse mackerel	3.37	3.37		3.61	1.83	1.83		1.83
Streaked gurnard	2.55	2.55		2.91	2.46	2.46		2.46
Pouting	3.02	3.02	2.47	3.28	2.27	2.27	2.46	2.33
Poor cod	2.45	2.45		2.83	2.38	2.38		2.38
John dory	3.58	3.58	2.96	3.80	2.46	2.46	2.65	2.52

C.6 Landing and discard species contribution per gear in the last 10, 5, and 3 years

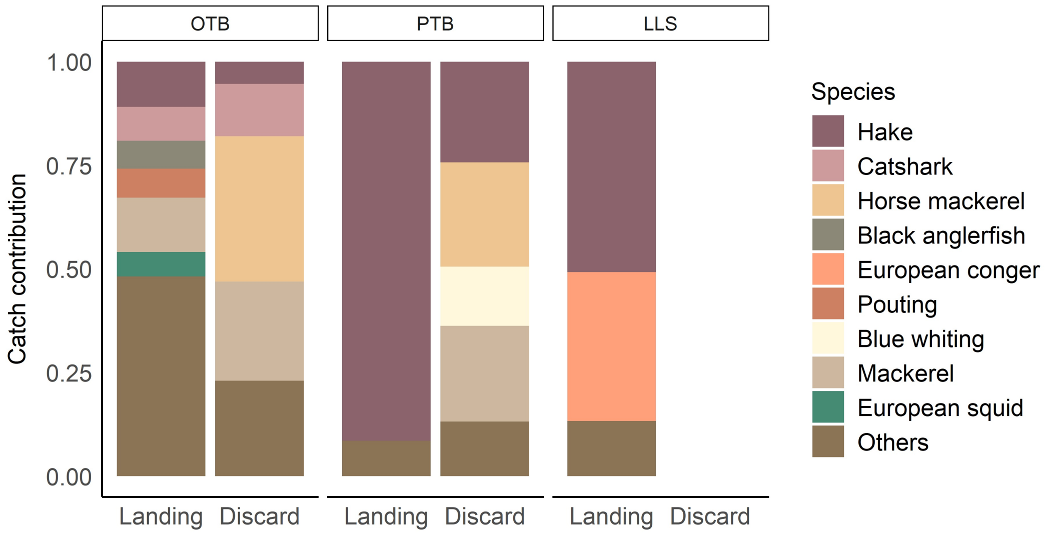


Figure C.1: Landings and discard species contribution (%) per gear in the last 10 years; bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). Species whose individual contribution was less than 5% of the total were grouped in "Others".

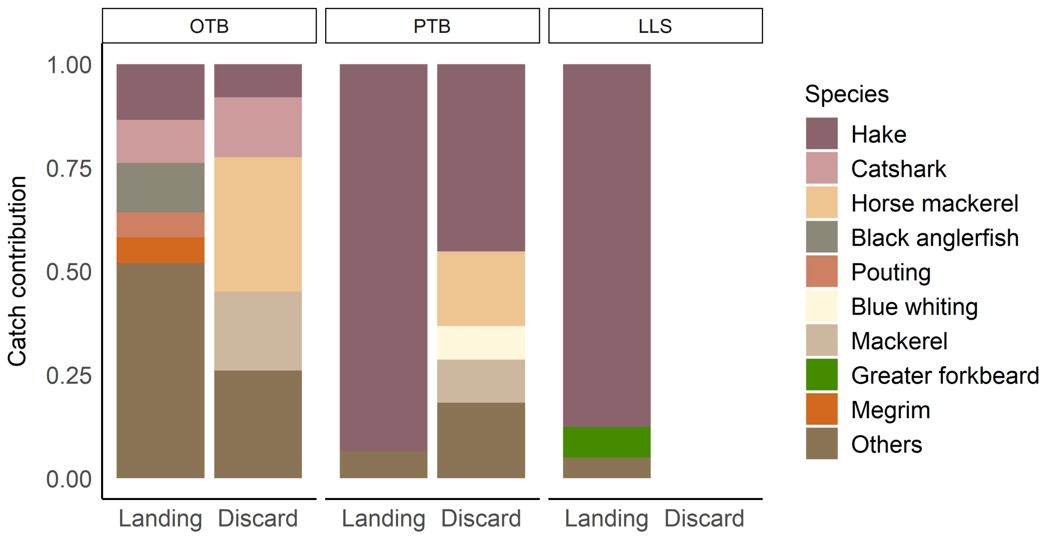


Figure C.2: Landings and discard species contribution (%) per gear in the last 5 years; bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). Species whose individual contribution was less than 5% of the total were grouped in "Others".

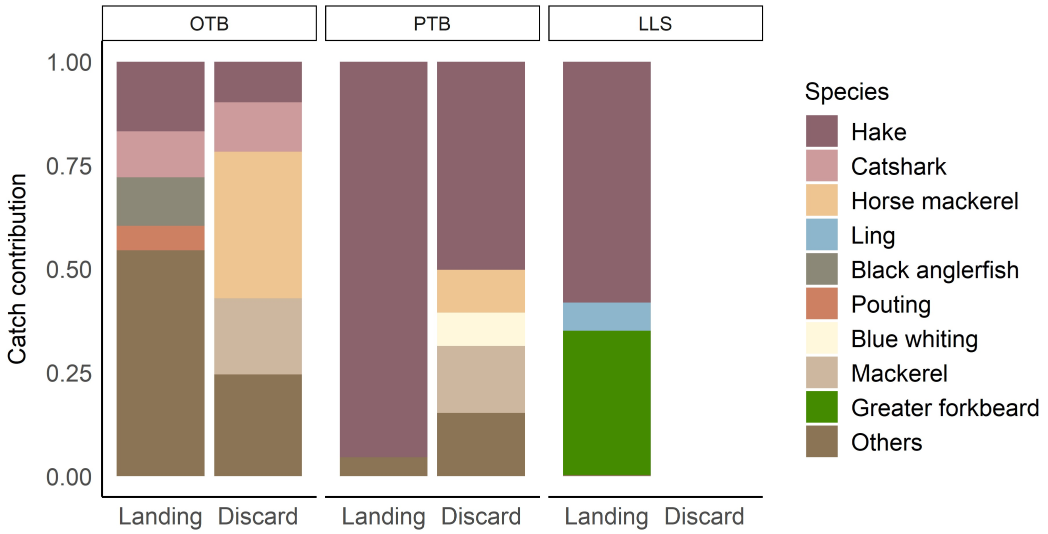


Figure C.3: Landings and discard species contribution (%) per gear in the last 3 years; bottom otter trawlers (OTB), bottom pair trawlers (PTB), and longliners (LLS). Species whose individual contribution was less than 5% of the total were grouped in “Others”.

C.7 Linear regressions for pair of productivity attributes

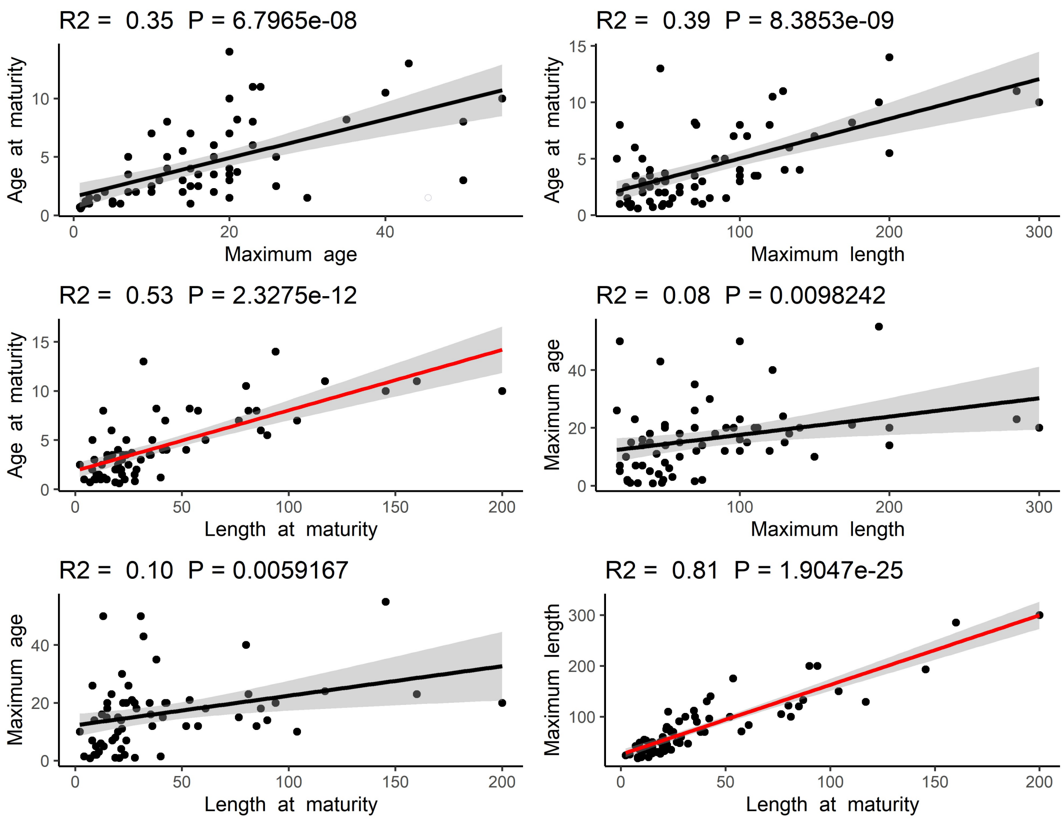
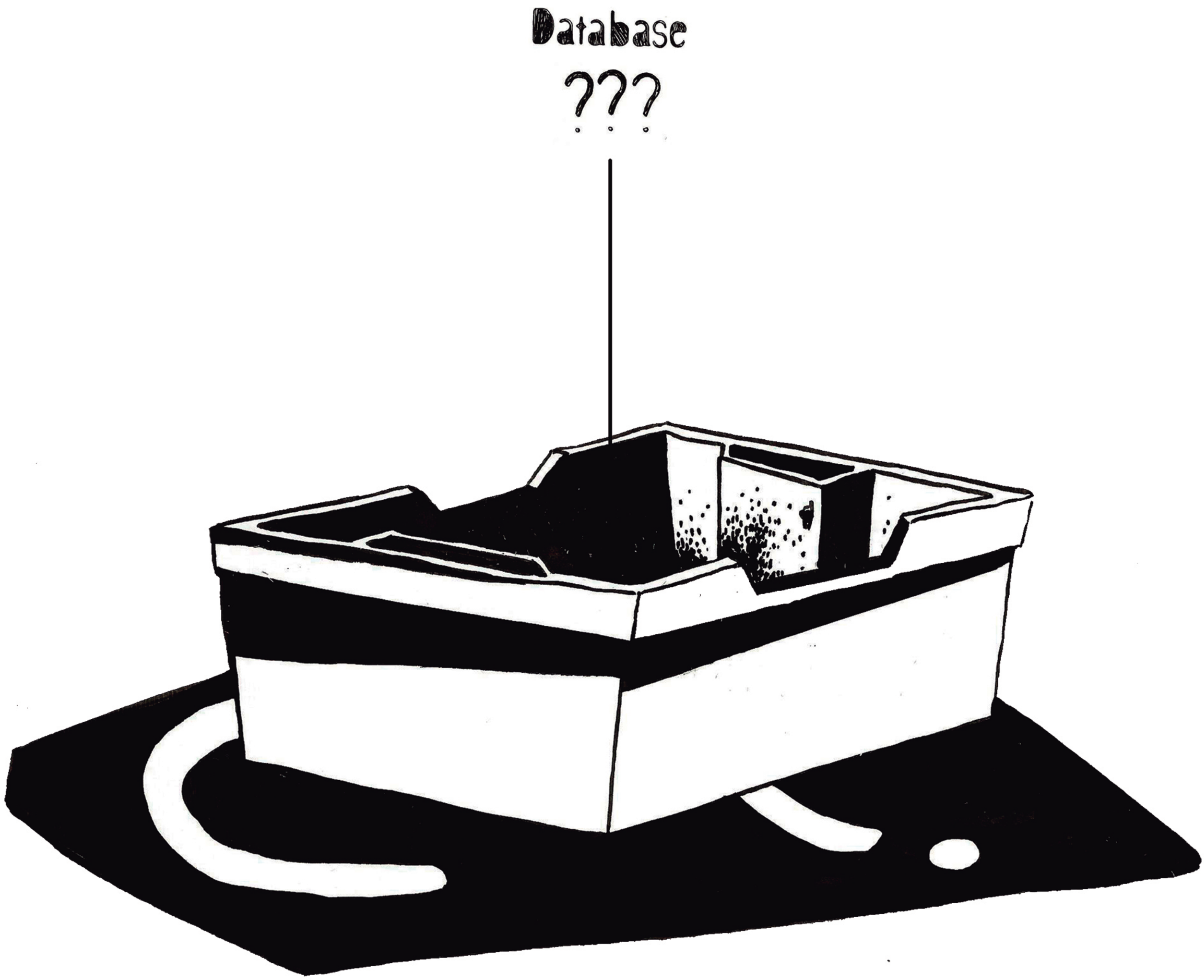


Figure C.4: Productivity attributes redundancy analysis. Linear regressions for pair of age at maturity, maximum age, length at maturity, and maximum length attributes. Red lines represent relationship with an adjusted R2 >0.5.



Appendix D

The value of data in SRA model

D.1 Introduction

Age-structured stock assessment models were first developed in the mid-20th century (Hilborn and Walters, 1992). Virtual Population Analysis (VPA) (Gulland, 1965; Pope, 1972) was the first age-structured assessment model used in fisheries management and is still in use today (Dichmont et al., 2016a). There are several variants of the VPA model that differ in the fitting of data and the algorithm employed to obtain a numerical solution (e.g., untuned VPA, ad hoc tuned VPA and statistically fit VPA). All these VPA variants require a time-series of catch-at-age data which is unavailable for many exploited stocks. As an alternative to the VPA approach, Kimura and Tagart (1982) developed the SRA model for situations where catch-at-age data are not available. The SRA model was further developed by Walters et al. (2006), incorporating Monte Carlo simulations to generate a probability distribution for stock dynamics, given alternative hypotheses regarding stock attributes (e.g., natural mortality or fishery selectivity). Kimura and Tagart (1982) and Walters et al. (2006) assumed that the stock was in unfished condition in the first year of the model's application. However, catch data may not be available from the beginning of a fish population's exploitation. Huynh et al. (2020a) recently developed a new implementation of the SRA model which allows the model to start from an exploited condition, accordingly assuming that the population in the first year of the model is in equilibrium with a catch level specified by the user, i.e., the estimated population in the first year of the model is able to produce this level of catch maintaining the population biomass constant (hereinafter "fished equilibrium assumption"). The fished equilibrium assumption as an initial population proxy is also used in other stock assessment models, for example in the widely used Stock Synthesis (Methot and Wetzel, 2013).

The most appropriate stock assessment model depends, among other factors, on the available data (Dichmont et al., 2021). Total catch and abundance index data respectively provide information on stock scale and trend; while length-structured and age-structured data supply information about stock scale and an improved understanding of population structure, natural mortality, growth and recruitment (Chen et al., 2003; Magnusson and Hilborn, 2007). Several studies have demonstrated that the assessment model estimates' accuracy varies depending on the data structure used (i.e., aggregated in biomass or by length or age) (Chen et al., 2003; Magnusson and Hilborn, 2007; Ono et al., 2014; Wetzel and Punt, 2011), length of the data time-series (Chen et al., 2003; Ono et al., 2014; Wetzel and Punt, 2011), distribution and sampling size of the data (Fisch and Bence, 2020; He et al., 2016; Hulson et al., 2017; Muradian et al., 2019), fishing mortality trajectory in the population (Magnusson and Hilborn, 2007; Ono et al., 2014) and the stock's biological parameters (Ono et al., 2014). Some of those studies assumed that the population was in an exploited condition in the first year of the data (Fisch and Bence, 2020; He et al., 2016; Hulson et al., 2017; Muradian et al., 2019). But the scenarios analysed in those studies included length-structured or age-structured information allowing the use of conventional assessment models to estimate the initial population. The remaining studies assumed that the population was in unfished condition in the first year of the assessment model (Chen et al., 2003; Ono et al., 2014; Wetzel and Punt, 2011). Understanding the impact of the initial population assumption on the assessment application is still an important challenge.

"Self-test" simulation is a fundamental analysis for assessing the performance of stock assessment models with respect to the ability to adequately estimate parameters (Deroba et al., 2014; Punt et al., 2020). Hordyk et al. (2019) evaluated the performance of the SRA model based on bias in biological parameters (natural mortality, age of maturity and steepness) and error in observation and model assumptions (hyper-stabilize or hyper-deplete abundance index, over-report or under-report catch and misspecification of selectivity curve shape). However, the performance of the SRA model has not been previously assessed in terms of utilizing different years of available data, different data structures, and alternative initial population assumptions.

The aim of this chapter is to evaluate by "self-test" simulation the SRA model's performance, depending on the lengths and structures of the time-series of catch and abundance index data and on the initial population assumption. The sensitivity of the results to various exploitation levels in the population and to alternative fishery selectivity assumptions is also analysed. The results allowed us to identify the data-limited stocks whose dynamics could be reliably simulated based on the SRA model.

D.2 Materials and Methods

The simulation framework to "self-test" the performance of SRA model consisted of generating a "true" population, adding observation error to the "true" population and fitting the assessment model to randomly generated data sets. Those datasets were generated under alternative data availability scenarios and different assumptions about initial population and fishery selectivity misspecification. The consistency between the "true" and estimated parameters was analysed using accuracy and bias performance statistics.

D.2.1 Simulation framework

The simulation algorithm was formed by the operating model (OM), the observation model and the assessment model. While OM represented the "true" population and exploitation dynamics, the observation model generated observations of catch and abundance indices from the OM. The assessment model consisted of applying the SRA model to the data generated by the observation model. Specifically, for each OM 100 population replicates were generated with varying recruitment deviations. The observed dataset of each population replicate was then produced with observation error. Finally, the SRA model was fitted to each observed dataset and 100 estimates were generated.

The simulation framework was based on the assessment of hake (*Merluccius merluccius*) distributed in ICES subareas 4, 6 and 7 and in divisions 3a and 8abd (ICES, 2019a). The stock assessment is conducted using Stock Synthesis (ICES, 2019a), which is an integrated stock assessment model (Methot and Wetzel, 2013). In this study, the OM and the observation model spanned the entire assessment model period (from 1978 to 2018, 41 years). The length of the data time-series was then subset to shorter periods when alternative length of the data time-series were used in the SRA model application to assess the effect of length in the performance of the SRA model.

D.2.1.1 Operating model

Four age-structured OMs were generated. In the first population, named $OM_{F_{SS3}}$, the fishing mortality (from 1978 to 2018) and the initial population in 1978 was equal to what was estimated by the stock assessment model for hake (ICES, 2019a). In this OM, the population in the first year was in exploited condition. In the other three OMs, the population started from an unfished condition, and different fishing mortality trajectories (Figure D.1) which lead to different depletion in the last year of the simulation were used. In these three populations, the annual fishing mortality (year^{-1}) increased linearly in the first 20 years and then remained constant for the following 21 years (Figure D.1). The fishing mortality in the most recent years was proportional to the fishing mortality at MSY for the stock ($\mathrm{F_{MSY}} = 0.26$) (ICES, 2019a). The three multipliers of $\mathrm{F_{MSY}}$ were 2, 1, and 0.5, which respectively corresponded to the $\mathrm{OM_{2F_{MSY}}}$, $\mathrm{OM_{F_{MSY}}}$ and $\mathrm{OM_{0.5F_{MSY}}}$ populations.

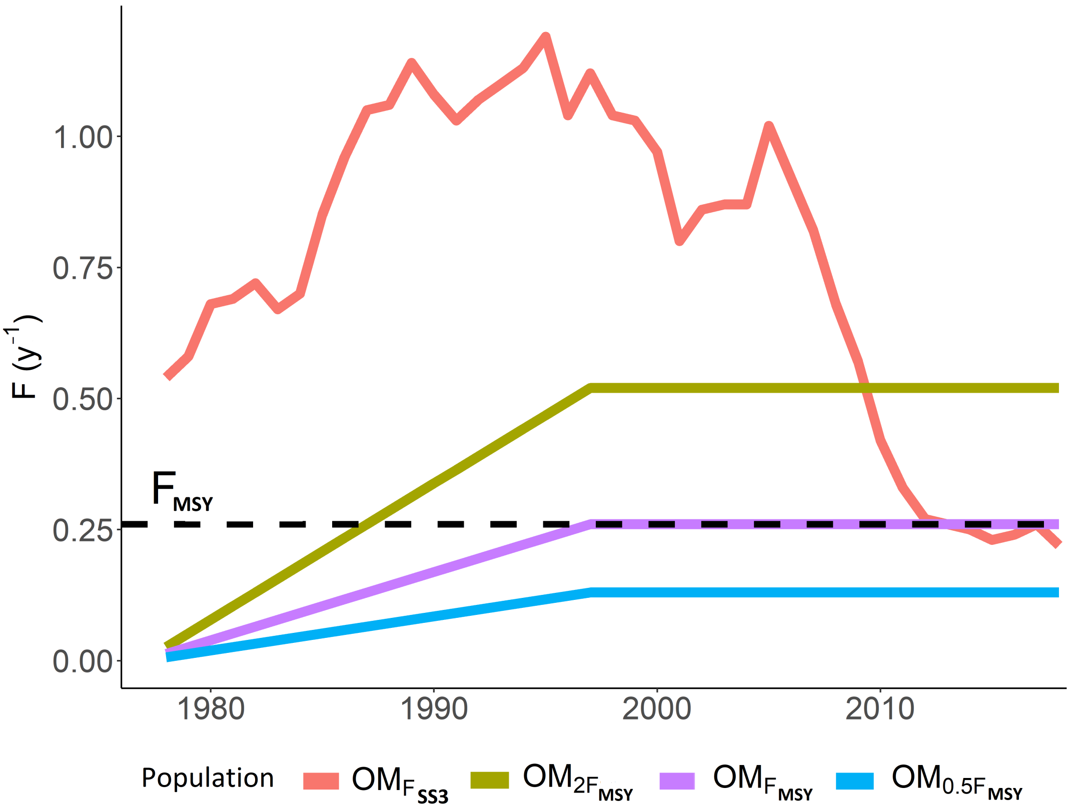


Figure D.1: Fishing mortality (F) in the $\mathrm{OM_{F_{SS3}}}$, $\mathrm{OM_{2F_{MSY}}}$, $\mathrm{OM_{F_{MSY}}}$ and $\mathrm{OM_{0.5F_{MSY}}}$ populations. The black horizontal dashed line represents fishing mortality at maximum sustainable yield ($\mathrm{F_{MSY}}$) for hake (ICES, 2019a).

In $\mathrm{OM_{2F_{MSY}}}$, $\mathrm{OM_{F_{MSY}}}$ and $\mathrm{OM_{0.5F_{MSY}}}$ populations, the following equation was used to calculate the initial population:

$$N_{1,a} = \begin{cases} R_0 & a = 1, \\ R_0\ e^{-M(a-1)} & a = 2 \dots A-1, \\ \frac{R_0\ e^{-M(A-1)}}{1-e^{-M}} & a = A, \end{cases} \tag{D.1}$$

where $N_{1,a}$ is the number of individuals-at-age a in the first year of the data ($y = 1$ corresponding to 1978), R_0 is the unfished recruitment, M is the natural mortality and A is the plus group representing all the individuals of age A and older. R_0, M and A were fixed at those values used in the stock assessment (ICES, 2019a) (Table D.1).

Table D.1: Life-history parameters of hake used to condition the operating model and to carry out the stock assessment of this study. Age range, von Bertalanffy growth parameters (L_∞ asymptotic length, k slope, and t_0 initial size), Beverton and Holt stock-recruitment relationship parameters (h steepness, R_0 recruitment in unfished condition, and σ standard deviation of the recruitment deviates), coefficient of variation of mean length-at-age data (cv), length-weight relationship parameters (a and b), maturity-at-age, and natural mortality (M).

Hake		**Value***
Age range		1-15⁺
Growth	L_∞ (cm)	130
	k (year^{-1})	0.177319
	t_0 (year)	0
Recruitment	h	0.96
	R_0 (thousands)	273,569.92
	σ	0.4
Length (cm) -at-age	cv	0.15
Length (cm) -weight (kg)	a	5.13E-6
	b	3.074
Maturity-at-age	1,2,3,4,5,>6 age	0.01, 0.34, 0.80, 0.95, 0.98, 1
Natural mortality	M (year^{-1})	0.4

* These values were taken from ICES (2019a). In ICES (2019a) the assessment runs from age 0 to age 15, where the recruitment happens at age 0 and age 15 is the plus group. In SRA the recruitment occurs at age 1. The age range affects the R_0 and h. Consequently, the ICES (2019a) values for R_0 (361,272 thousands) and h (0.99) were recalculated using Beverton and Hold stock-recruitment relationship equation (Beverton and Holt, 1957) before their use in the OM and assessment model.

In the four populations, the population numbers-at-age for the subsequent years ($N_{y,a}$) were given by the age-structured exponential survival equation (Quinn and Deriso, 1999):

$$N_{y,a} = \begin{cases} R_y & a = 1, \\ N_{y-1,a-1}e^{-Z_{y-1,a-1}} & a = 2, \ldots, A-1, \\ N_{y-1,a-1}e^{-Z_{y-1,a-1}} + N_{y-1,a}e^{-Z_{y-1,a}} & a = A, \end{cases} \tag{D.2}$$

where R_y is the annual recruitment and $Z_{y,a}$ is the total mortality calculated as the sum of M and the annual fishing mortality-at-age ($F_{y,a}$). In all the equations subscripts y and a correspond to year and age respectively.

Recruitment (R_y) occurred at age 1 and was modelled using the Beverton and Holt stock-recruitment relationship (Beverton and Holt, 1957):

$$R_y = \frac{4 * h * R_0 * SSB_{y-1}}{(SSB_0 * (1-h)) + (SSB_{y-1} * (5 * h - 1))} * exp\left(\delta_y - 0.5 * \sigma^2\right), \delta_y \sim N\left(0; \sigma^2\right), \quad \text{(D.3)}$$

where SSB_y is the annual spawning stock biomass, SSB_0 is the spawning stock biomass in unfished condition, h is the steepness (the proportion of R_0 when the SSB is reduced to 0.2 the SSB_0), δ_y is the annual recruitment deviation and σ is the standard deviation of the recruitment deviates.

For each OM 100 population replicates were generated sampling δ_y. δ_y follows a normal distribution with mean equal to 0 and a standard deviation equal to σ. h, R_0 and σ were fixed to the value used in the assessment of hake (ICES, 2019a) given in Table D.1. SSB_0 was defined based on the values of R_0, length-weight relationship parameters (a and b) and the proportion of mature individuals-at-age (Table D.1).

The fishing mortality-at-age a in year y ($F_{y,a}$) was calculated as the product of the fishing mortality by year (Figure D.1) and the annual fishery selectivity-at-age (Figure D.2). The annual fishery selectivity-at-age used to generate the four populations was fixed to the overall fishery selectivity-at-age estimated by ICES (2019a). In the first eight years (from 1978 to 1986) the fleet mostly caught age 1 individuals, whereas in the most recent years (from 1987 to 2018) the maximum fishery selectivity-at-age was at age 4 (Figure D.2).

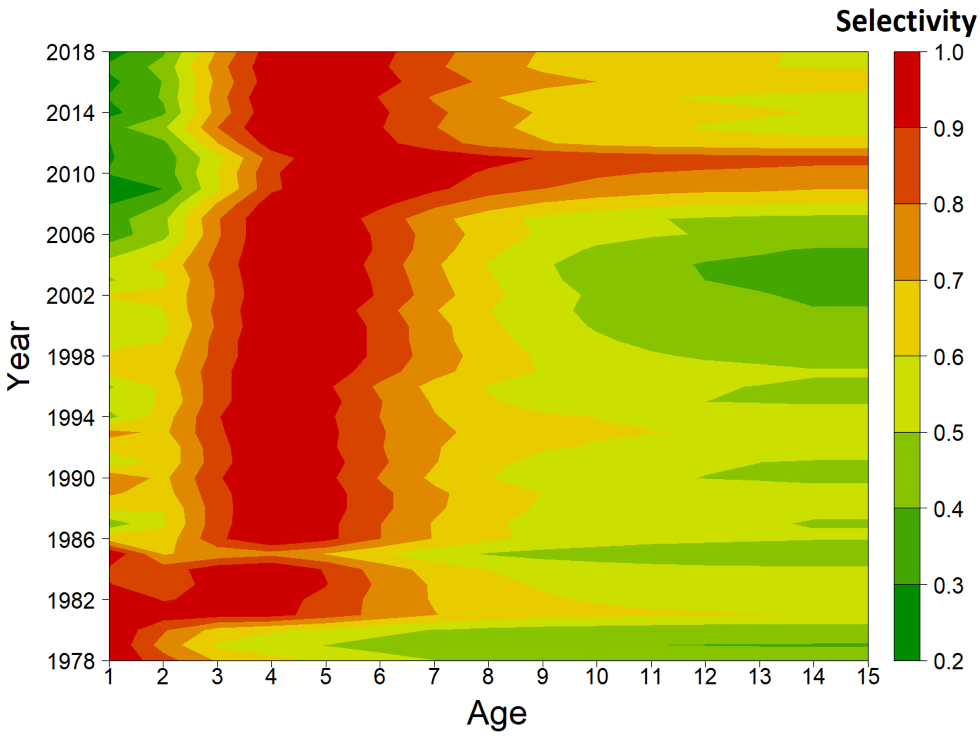


Figure D.2: Fleet selectivity-at-age of hake from 1978 to 2018.

Annual catch-at-age in the "true" populations were calculated using the Baranov catch equation (Baranov, 1918). Annual total catch in biomass was derived from the sum of products of catch-at-age and weight-at-age derived from biological parameters in Table D.1. As the OMs were age-structured, the catch-at-length of the "true" populations were generated as the product of catch-at-age and the probability of having length l, given age a, $P(l|a)$:

$$P(l|a) = \begin{cases} \Phi({L'}_{l+1}), & l = 1, \\ \Phi({L'}_{l+1}) - \Phi({L'}_l) & l = 2,\ \dots,\ L-1, \\ 1 - \Phi({L'}_l) & l = L, \end{cases} \tag{D.4}$$

where L'_l is the length at the lower boundary of the length bin l. Length bins ranged from 5.5 to 135.5 in intervals of 2 cm. $\Phi({L'}_l)$ is the cumulative distribution function of a normal variable with mean equal to the mean length-at-age and a standard deviation which is the product of mean length-at-age and the coefficient of variation in mean length-at-age.

The mean length-at-age was calculated using von Bertalanffy growth model parameters and the coefficient of variation of mean length-at-age data given in Table D.1 which are the ones used by the assessment working group (ICES, 2019a). The "true" mean length in the catch data were equal to the weighted mean of the catch-at-length data.

D.2.1.2 Observation model

Observation errors were introduced in the "true" populations data to generate the observed data. The observed total catch was generated by multiplying the "true" total catch with a lognormal error sampled with mean equal to 1 and a coefficient of variation equal to 0.25. The observed biomass index was created with a lognormal distribution with mean equal to the biomass of the population at the beginning of the year and a coefficient of variation of 0.25. The proportion of individuals observed in each age or length class was modelled using a multinomial distribution with an Effective Sample Size (ESS) of 50 for proportions at-age, as done in Magnusson and Hilborn (2007), and of 125 for proportions at-length, as used by default in the stock assessment (ICES, 2019a). The general formula used to obtain observed catch-at-length, catch-at-age and index-at-age data ($\widehat{X_{y,s}}$) is:

$$\widehat{X_{y,s}} \ \sim \ Multinomial\ (ESS,\ P(s=1,y),\ \dots\ ,\ P(s=S,y))\ *\ \frac{X_y}{ESS}, \tag{D.5}$$

where $\widehat{X_{y,s}}$ is the observed catch-at-length, observed catch-at-age or observed index-at-age, s is the age or length bin, S is the maximum age or length bin, $P(s,y)$ is the proportion of individuals in bin s in year y, and X_y is the "true" number of individuals during year y. The proportions were taken from the "true" catch-at-length, catch-at-age and index-at-age data. The observed mean length in the catch was created with a lognormal distribution with mean equal to the weighted mean of the "true" catch-at-length data, and a coefficient of variation of 0.25.

D.2.1.3 Assessment model

The assessment model used to obtain population abundance and exploitation level estimates was the SRA model available in the MSEtool package version 2.0.1 (Huynh et al., 2020a) within the R environment (R Core Team, 2022). The population dynamics in the SRA model is based on the same equations used in the OM – the exponential survival equation, the Beverton and Holt stock-recruitment relationship equation and the Baranov catch equation (Walters et al., 2006). Also, the SRA model uses the same equations as those used in the OM to predict the mean catch length, catch-at-length, catch-at-age and index-at-age data (Huynh et al., 2020a). In the SRA model, unless otherwise specified, fishery selectivity is constant over time. When the fishery selectivity is unknown, the fishery's historical selectivity is inferred from age- or length-structured catch data. Without age- or length-structured data, the age dependent fishery selectivity (sel_a) is defined as follows:

$$sel_a = \begin{cases} 2^{-[(L_a \; - \; L_{FS})/\sigma_{asc}]^2} & L_a \; < \; L_{FS}, \\ \\ 2^{-[(L_a \; - \; L_{FS})/\sigma_{des}]^2} & L_a \; \geq \; L_{FS}, \end{cases} \tag{D.6}$$

where $\sigma_{asc} = (L_5 - L_{FS})/\sqrt{-log_2\,(0.05)}$ and $\sigma_{des} = (L_\infty - L_{FS})/\sqrt{-log_2\,(V_{maxlen})}$ control the shape of the ascending and descending limbs of the fishery selectivity curve, respectively. L_a is the mean length-at-age, L_∞ is the asymptotic length, L_5 is the length of 5% selectivity, L_{FS} is the length of full selectivity, and V_{maxlen} is the selectivity at L_∞.

The SRA model consists of finding the combination of initial biomass, fishing mortality and recruitment levels that could have generated the observed data given a specific set of values for biological parameters (von Bertalanffy growth parameters, steepness, length-weight relationship parameters, maturity-at-age and natural mortality), standard deviation of the recruitment deviates, and survey and fishery selectivities. The minimum data required to apply the SRA model is annual total catch and total abundance index data. Other additional data, such as annual mean length in the catch, catch-at-length, catch-at-age and index-at-age data can also be supplied to the SRA model. Maximum log-likelihood is used by the SRA to estimate the fishing mortality, fishery selectivity, R_0, recruitment deviations and the initial equilibrium fishing mortality. The catchability coefficient (q) is solved analytically. The model equations to estimate parameters are presented in Appendix E.1. The maximum log-likelihood method consists of maximizing the sum of the log-likelihoods for recruitment deviation, fished equilibrium assumption, total catch data, abundance index data and any other additional dataset used. The equations used to estimate the log-likelihood components are presented in Appendix E.2. The SRA model allows the inclusion of uncertainty in the log-likelihood components of all the input data. As an exception, the coefficient of variation of the total catch data and fished equilibrium assumption is fixed at 0.01, assuming low error in the total catch data and in the fished equilibrium assumption.

When applying the SRA model, the biological parameters and the standard deviation of the recruitment deviates were assumed to be known without error (Table D.1) and the values were taken from the OM. The observed biomass index data were proportional to population biomass, so survey selectivity was equal to 1 for all ages and years and it was fixed at 1 in the SRA application. The annual fishery selectivity was assumed to be known without error in all the scenarios (Figure D.2), except in those scenarios where the sensitivity of the results to fishery selectivity misspecification was analysed.

The uncertainty of the parameters in the SRA model application were also set at the "true" values. The coefficient of variation of the biomass index data and mean length catch data were fixed at 0.25, as used in the generation of biomass index data and mean length catch data in the observation model. The distributions of age and length classes in catch-at-length, catch-at-age and index-at-age data were defined with the same effective sample size used in the generation of that data in the observation model. For length-age conversion, the coefficient of variation in mean length-at-age was fixed at 0.15, as used in the observation model.

The version of the SRA model used in this study was specifically designed to assess populations for which the catch data do not start at the same time as the fishing activity. When the population starts from an exploited condition, the SRA model estimates the unfished recruitment and the initial equilibrium fishing mortality based on the user-specified initial catch level and stock life-history parameters (e.g., stock-recruitment relationship and natural mortality). From these estimates of equilibrium fishing mortality, the initial depletion is derived (Huynh et al., 2020a). When the population starts from unfished condition, the initial depletion is known (i.e., equal to 0) and there are less parameters to estimate. Applying the assessment model to unfished population is thus the best situation for SRA or very likely for any other assessment model to obtain accurate results (Huynh et al., 2020a,b).

Scenarios

The SRA model's performance was studied in 24 data availability cases that corresponded to the combination of six data type and four data time-series length scenarios. All the cases included the total catch and biomass index data, which is the minimum data required to run the SRA model. In one data type scenario, we did not include any additional data (CI scenario); in the other five scenarios, we included the following additional data: mean length catch data (ML scenario), catch-at-length data (CAL scenario), catch-at-age data (CAA scenario), index-at-age data (IAA scenario) and both catch-at-age and index-at-age data (CAA+IAA scenario). The four data time-series length scenarios started in 1978 and finished in either 2018 (41 years), 2009 (32 years), 1999 (22 years) or 1989 (12 years). So the differences found in alternative data time-series length scenarios will be due to the length of the data time-series rather than the assumed initial population in the first year of the data.

The data availability scenarios were analysed for two initial population situations: (1) when the population started from unfished condition and the catch data were assumed to be known since the beginning of the exploitation, and (2) the catch data were assumed to start later than the fishing activity and the population in the first year of the data was in unknown exploited conditions. For the first, the $\mathrm{OM}_{2\mathrm{F}_{\mathrm{MSY}}}$, $\mathrm{OM}_{\mathrm{F}_{\mathrm{MSY}}}$ and $\mathrm{OM}_{0.5\mathrm{F}_{\mathrm{MSY}}}$ populations were used. For the second, the population based on hake fishing mortality trajectory, the $\mathrm{OM}_{\mathrm{F}_{\mathrm{SS3}}}$ population, was used. In the assessment of hake (ICES, 2019a), a catch level of 53,564 tonnes, which is the mean of the first five years' catch (from 1978 to 1982), is used as fished equilibrium assumption when applying the Stock Synthesis assessment model. In this chapter, results for 50,000 tonnes as fished equilibrium assumption were studied in detail, and a range of fished equilibrium assumptions from 40,000 to 70,000 tonnes was used to analyse the sensitivity of the results to the assumed equilibrium catch level.

Finally, the impact of the fishery selectivity misspecification in the SRA model was studied. To that end, the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used, and in 24 data availability cases the results' accuracy with respect to alternative fishery selectivity settings was explored. In those scenarios where fishery age- or length-structured data were assumed to be known (i.e., CAL, CAA and CAA+IAA scenarios), three alternative settings to define fishery selectivity were analysed. In one selectivity scenario, the three fishery selectivity parameters (L_5, L_{FS} and V_{maxlen}) were estimated from age- or length-structured data (Estimated scenario). In the other two selectivity scenarios, the L_5 and L_{FS} selectivity parameters were estimated from age- or length-structured data but the V_{maxlen} parameter was fixed to the values in Table D.2 and to 1, assuming in the latter the logistic selectivity curve shape (Fixed V_{maxlen} and Logistic scenarios, respectively). In those scenarios where fishery age- or length-structured data were assumed to be unknown (i.e., CI, ML and IAA scenarios), two alternative selectivity misspecification scenarios were analysed. In the first selectivity scenario, the parameters of the selectivity curve were uniformly sampled from a defined range of estimated values (Table D.2) (Sampled scenario). In the second selectivity scenario, the V_{maxlen} parameter was set at 1 and the other two selectivity parameters were uniformly sampled from a defined range of values (Table D.2) (Logistic scenario). The range of each selectivity parameter was defined assuming a coefficient of variation of 10% in a uniform distribution centered on the estimated value. The three selectivity parameters were estimated for each data time-series length scenario by adjusting the selectivity curve to the observed curves in the "true" population.

Table D.2: Estimated fleet selectivity parameters from the "true" population and the lower and upper limit of the estimates in a uniform distribution with a coefficient of variation of 10%. Selectivity parameters: the length of 5% selectivity (L_5), the length of full selectivity (L_{FS}) and the selectivity at asymptotic length (V_{maxlen}). Data time-series length scenarios (TS): 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

	Selectivity parameters								
	L_5			L_{FS}			V_{maxlen}		
TS	**Lower**	**Estimated**	**Upper**	**Lower**	**Estimated**	**Upper**	**Lower**	**Estimated**	**Upper**
41	10.77	13.03	15.29	53.38	64.56	75.74	0.27	0.33	0.39
32	3.99	4.82	5.66	49.66	60.07	70.47	0.25	0.31	0.36
22	0.48	0.58	0.68	41.10	49.71	58.32	0.28	0.34	0.40
12	4.85	5.86	6.88	21.25	25.70	30.15	0.31	0.37	0.44

The list of parameters estimated in each scenario is summarized in Table D.3.

Table D.3: List of the parameters estimated depending on the fishery selectivity (Sel), population (Pop), data type (DT), data time-series length (TS) and fished equilibrium assumption (Feq) scenario combination. Selectivity scenarios: yearly selectivity was fixed to the "true" values (True), the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen}), the selectivity parameters were estimated (Estimated), the logistic selectivity was assumed (Logistic) and the selectivity parameters were sampled from a range of values uniformly (Sampled). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

Scenarios					Estimated parameters							
					F_y	Sel param			R_0	δ_y	Feq	q
Sel	Pop	DT	TS	Feq		L_5	L_{FS}	V_{maxlen}				
True	$OM_{2F_{MSY}}$	CI,ML,CAL, CAA,IAA, CAA+IAA	41, 32, 22, 12	Unfished	x				x	x		x
	$OM_{F_{MSY}}$				x				x	x		x
	$OM_{0.5F_{MSY}}$				x				x	x		x
	$OM_{F_{SS3}}$			40,000-70,000	x				x	x	x	x
Fixed V_{maxlen}		CAL,CAA, CAA+IAA		50,000	x	x	x		x	x	x	x
Estimated					x	x	x	x	x	x	x	x
Logistic					x	x	x		x	x	x	x
		CI,ML,IAA			x				x	x	x	x
Sampled					x				x	x	x	x

D.2.2 Key estimates

The influence of alternative data type, data time-series length and initial population assumption on the results was assessed using four estimates (hereinafter "key estimates") selected due to their potential interest for management: depletion (D, one minus the ratio of the last year spawning stock biomass and the unfished condition spawning stock biomass), unfished recruitment (R_0), last year spawning stock biomass (SSB_{LY}) and last year fishing mortality (F_{LY}). Furthermore, the influence of the initial population assumption on the results was also assessed by analysing the estimates of the initial depletion (D_{init}) and the time-series of SSB. Initial depletion values range from 0 to 1, where 0 means that the population was in unfished condition and 1 means that the population was extinct.

D.2.3 Performance statistics

The performance of the SRA model was assessed using accuracy and bias statistics of the key estimates (Ono et al., 2014; Walther and Moore, 2005). Only the converged SRA model fits were selected to calculate the performance statistics. Median values across all the population replicates were used rather than the mean values to make the performance statistics more robust to outliers.

For each population replicate i, let θ_i denote the "true" value of the key estimates in the simulated OM, and $\widehat{\theta}_i$ denote the estimated value from the SRA (i.e., the median of the 100 SRA model fit estimates). The accuracy of each estimate was measured in terms of the median absolute relative error (MARE) and the failure rate (FR). The median absolute relative error (MARE) is the median across all the population replicates absolute relative error (ARE), and it was calculated as:

$$MARE = \ Median \ \left(\left| \frac{\widehat{\theta}_1 - \theta_1}{\theta_1} \right|, \ \ldots, \left| \frac{\widehat{\theta}_{100} - \ \theta_{100}}{\theta_{100}} \right| \ \right) * 100, \tag{D.7}$$

where the higher the value of MARE the lower the accuracy of the key estimates. The MARE values enable evaluation of relative differences in accuracy of the key estimates.

As in Magnusson and Hilborn (2007), the FR was defined as the proportion of the estimates that were less than half or greater than twice the "true" value, mathematically:

$$FR \ = Pr \left(\left| \widehat{\theta}/\theta \right| \ < \ 0.5 \ and \ \left| \widehat{\theta}/\theta \right| \ > 2 \right). \tag{D.8}$$

The range of values two times higher or two times lower than the "true" value may often be undetectable in most real situation. This performance statistic, FR, enables us to evaluate the results, allowing a moderately low error in the key estimates' accuracy.

The bias was calculated using the median relative error (MRE) of all the population replicates as follows:

$$MRE = \ Median \ \left(\frac{\widehat{\theta}_1 - \ \theta_1 .}{\theta_1}, \ \ldots, \frac{\widehat{\theta}_{100} - \ \theta_{100}}{\theta_{100}} \right) * 100, \tag{D.9}$$

where the higher the absolute value of MRE the higher the bias in the estimates.

D.3 Results

The SRA model's convergence rate was above 80% in all the scenarios, reaching 100% in most cases. Notably, the convergence rate was approximately 17% in scenarios where mean length catch data (ML scenario) with the longest data time-series and misspecified selectivity (Sampled and Logistic scenarios) were used. The convergence rates of each scenario are presented in Table E.1 in Appendix E.3.

D.3.1 Unfished condition

The key estimates were more accurate (i.e., lower FR and MARE) when a longer data time-series was used. In all the data type scenarios the FR and MARE of the key estimates increased when the length of the data time-series was shortened (Table D.4, Figure D.3). The range in absolute relative error (ARE) also increased when shorter length of data time-series was used (Figure E.2 in Appendix E.4). The accuracy of the key estimates was high (FR lower than 20%) for all data time-series length and data type scenario combinations except with the shortest data time-series (12 years), where the FR reached values up to 36% and MARE up to 50%.

In the CI scenario, less data than in the rest of the scenarios was used and withing each data time-series length scenario, it had the poorest estimates of depletion, R_0, SSB_{LY} and F_{LY} in terms of accuracy (they had the highest values of FR and MARE) (Table D.4). The accuracy of the key estimates was higher with catch-at-length data (CAL scenario) than with mean length catch data (ML scenario) (lower MARE in the former). The accuracy of the key estimates was similar among scenarios where catch-at-length, catch-at-age or index-at-age data were used (CAL, CAA and IAA scenarios respectively). In general, the most accurate key estimates were obtained with the most complete data (CAA+IAA scenario).

When increasing the length of the data time-series, the accuracy of SSB_{LY} and F_{LY} increased more than that of depletion and R_0 estimates (Table D.4). In all the cases, the estimates of depletion and R_0 were more accurate (i.e., lower FR and MARE) than the estimates of SSB_{LY} and F_{LY}, with the estimates of F_{LY} the least accurate. In the CI scenario, the most accurate key estimate was depletion. In the ML, CAL, CAA, IAA and CAA+IAA scenarios, the key estimate with the lowest MARE was R_0, except with the shortest data time-series, where the accuracy of depletion was higher than the accuracy of R_0. The estimates of depletion and R_0 were more unbiased (i.e., lower absolute values of RE) than the estimates of SSB_{LY} and F_{LY} (Table D.4, and Figure E.3 in Appendix E.4).

The accuracy of the key estimates depended on the fishing mortality of the populations (Figure D.3). The accuracy of R_0, SSB_{LY} and F_{LY} increased when fishing mortality in the population was higher. These key estimates had lower MARE in the $OM_{2F_{MSY}}$ population relative to the $OM_{F_{MSY}}$ population, and higher MARE in the $OM_{0.5F_{MSY}}$ population relative to the $OM_{F_{MSY}}$ population (Figure D.3). The estimates of depletion showed the opposite trend: the accuracy of depletion was lower in the $OM_{2F_{MSY}}$ population and higher in the $OM_{0.5F_{MSY}}$ population relative to the $OM_{F_{MSY}}$ population. The shorter the data time-series, the higher the differences between populations in terms of accuracy of R_0, SSB_{LY} and F_{LY}. On the contrary, the difference between the MARE of depletion in the $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ and $OM_{0.5F_{MSY}}$ populations was lower when the shortest length of the data time-series was used (Figure D.3).

Table D.4: The performance statistics of the key estimates for each data type and data time-series length scenario combination when the $OM_{F_{MSY}}$ population was used. Performance statistics: failure rate (FR), median absolute relative error (MARE), median relative error (MRE). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

		FR				**MARE**				**MRE**			
		12	**22**	**32**	**41**	**12**	**22**	**32**	**41**	**12**	**22**	**32**	**41**
D	**CI**	0	0	0	0	16	13	12	12	0	-1	4	1
	ML	0	0	0	0	15	12	12	12	-3	-7	0	-4
	CAL	0	0	0	0	11	11	9	10	-5	-7	-3	-7
	CAA	0	0	0	0	13	13	10	10	-1	0	3	0
	IAA	0	0	0	0	12	11	10	10	-1	-1	0	0
	CAA+IAA	0	0	0	0	12	11	9	8	-2	1	1	-1
$\mathbf{R_0}$	**CI**	20	5	0	0	33	17	19	17	27	15	19	17
	ML	12	0	0	0	29	11	12	11	7	8	11	11
	CAL	0	0	0	0	18	9	7	5	-12	-7	-4	-3
	CAA	13	0	0	0	27	9	7	6	3	5	5	5
	IAA	7	0	0	0	26	9	9	7	6	5	7	5
	CAA+IAA	8	0	0	0	22	9	7	6	4	5	5	4
$\mathbf{SSB_{LY}}$	**CI**	31	15	4	2	47	23	25	21	25	16	21	16
	ML	24	3	0	0	41	18	17	14	8	3	7	7
	CAL	11	0	0	0	24	17	12	12	-16	-13	-7	-8
	CAA	21	1	0	0	37	19	12	12	3	4	7	6
	IAA	20	0	2	0	41	17	12	13	1	4	6	4
	CAA+IAA	20	0	0	0	33	15	11	12	2	4	4	4
$\mathbf{F_{LY}}$	**CI**	36	19	17	7	50	34	31	27	-23	-16	-17	-17
	ML	32	12	10	3	45	31	27	20	-10	-8	-7	-9
	CAL	21	10	6	2	36	30	24	24	15	12	8	8
	CAA	34	8	8	4	44	24	27	24	-13	-4	-5	-7
	IAA	28	6	10	4	43	29	25	22	-11	-4	-9	-4
	CAA+IAA	27	4	9	5	43	24	23	22	-7	2	-6	-3

In general, the estimates of depletion were more accurate than those of R_0 when the exploitation level was low in the most recent years ($OM_{0.5F_{MSY}}$ population), and the estimates of R_0 were more accurate than depletion when the population was highly exploited ($OM_{2F_{MSY}}$ population) (Figure D.3).

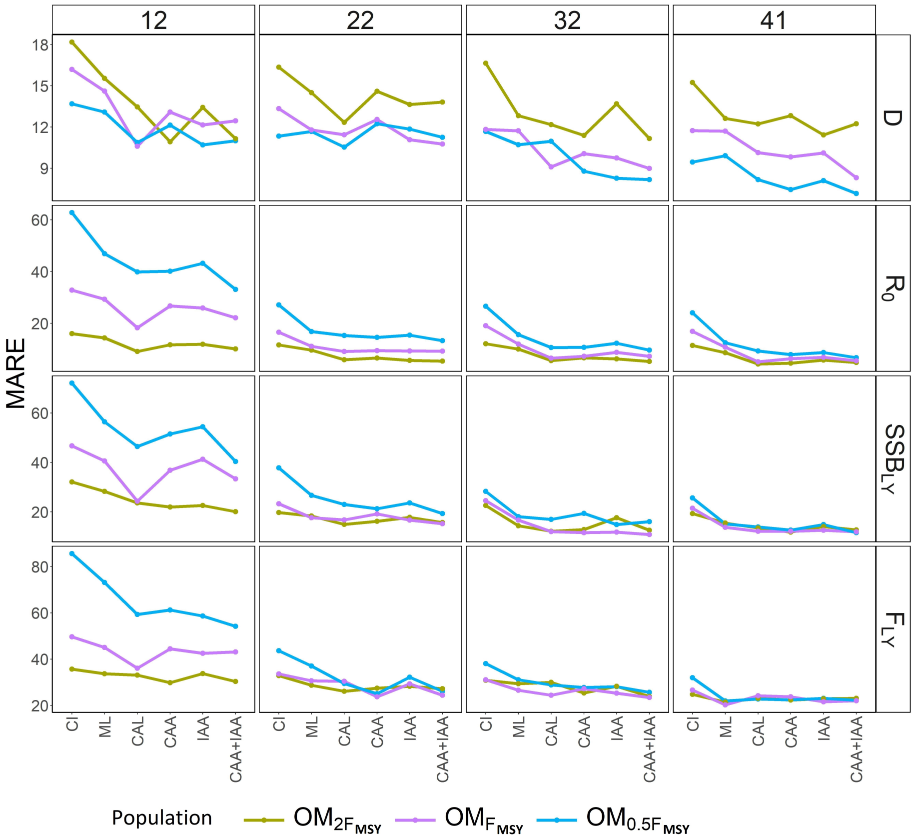


Figure D.3: The median absolute relative error (MARE) of the key estimates (in rows) for each data type (x axis) and data time-series length scenario (in columns) combination when the $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ and $OM_{0.5F_{MSY}}$ populations were used. Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

D.3.2 Exploited condition

For the $OM_{F_{SS3}}$ population the longest data time-series also gave the most accurate estimates. The worst performance was obtained with the shortest data time-series (Table D.5, and Figure E.4 and Figure E.5 in Appendix E.5). The influence of the length of the data time-series on the key estimates' accuracy was high when the minimum required data (CI scenario) was used (Table D.5). The CI scenario gave accurate and unbiased key estimates (i.e., FR lower than 10%, and MRE between -8 and 6) when the longest data time-series (41 years) was used. However, with a shorter data time-series length (i.e., when 32, 22 and 12 year lengths of data were used), the FR of depletion, SSB_{LY} and F_{LY} was higher than 40%. This means that in over 40% of the simulations, the key estimates were two times higher or two times lower than the "true" value. With additional data (ML, CAL, CAA, IAA and CAA+IAA scenarios), in all the data time-series length scenarios, the accuracy of the key estimates was high (the FR of the key estimates was lower than 20%). In all the data type scenarios, the median bias of the estimates were close to 0 (Table D.5), and the "true" values of the key estimates were inside the confidence interval of 80% of the relative error (RE) (Figure E.6 in Appendix E.5).

Table D.5: The performance statistics of the key estimates for each data type (in rows) and data time-series length (in columns) scenario combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Performance statistics: failure rate (FR), median absolute relative error (MARE), and median relative error (MRE). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

		FR				**MARE**				**MRE**			
		12	**22**	**32**	**41**	**12**	**22**	**32**	**41**	**12**	**22**	**32**	**41**
D	**CI**	72	70	62	4	325	203	130	24	325	203	130	3
	ML	14	8	3	0	31	29	17	17	-10	-15	-15	-11
	CAL	0	0	0	0	19	18	13	12	-7	-7	-8	-5
	CAA	3	1	0	0	23	18	15	12	13	7	3	1
	IAA	1	1	0	0	21	19	16	10	9	9	3	2
	CAA+IAA	0	1	0	0	19	17	14	11	9	6	2	3
R_0	**CI**	18	16	7	0	25	25	24	13	-11	-20	-21	6
	ML	3	2	0	0	19	19	18	15	9	14	17	15
	CAL	0	0	0	0	9	10	9	8	1	4	4	3
	CAA	0	0	0	0	12	9	7	6	-3	-2	-2	0
	IAA	0	0	0	0	9	9	8	7	-1	1	1	1
	CAA+IAA	0	0	0	0	11	8	7	6	-4	-1	-1	0
SSB_{LY}	**CI**	64	56	42	1	225	138	76	13	225	138	76	5
	ML	7	2	1	0	21	19	14	12	5	-2	-1	0
	CAL	0	0	0	0	17	14	12	11	-2	-3	-2	-1
	CAA	0	0	0	0	18	15	15	9	13	5	4	2
	IAA	1	0	0	0	21	18	15	10	5	7	5	3
	CAA+IAA	0	0	0	0	17	15	14	11	8	5	6	2
F_{LY}	**CI**	66	57	56	3	77	63	55	24	-72	-61	-52	-8
	ML	20	15	13	2	29	33	27	19	-15	1	1	-2
	CAL	17	12	11	2	31	30	26	21	-5	8	-2	-3
	CAA	15	13	10	1	32	34	25	20	-22	-3	-8	-2
	IAA	14	12	10	1	35	34	25	20	-16	-6	-10	-5
	CAA+IAA	13	13	9	1	30	34	23	20	-18	-6	-6	-4

In the scenario where the mean length catch data were used (ML scenario) and in the scenarios where length-structured or age-structured data were used (CAL, CAA, IAA and CAA+IAA scenarios) the accuracy of depletion, SSB_{LY} and F_{LY} was more similar across scenarios than the accuracy of R_0. The MARE value of R_0 was around two times higher when mean length in catch data were used than when length-structured or age-structured data were used (Table D.5). The same happened with the bias: the estimates of R_0 were more than four times biased (the absolute MRE value of R_0 was around four times higher) in the ML scenario than in scenarios where length-structured or age-structured data were used (Table D.5).

D.3.2.1 Sensitivity analysis of the key estimates to the fished equilibrium assumption

In any data availability scenario, the estimates of R_0, SSB, depletion and initial depletion increased when higher catch as fished equilibrium assumption was used (Figure D.4, Figure D.5). As an exception, when the minimum required data with the longest data time-series length was used, the estimates of SSB obtained with the lowest catch as fished equilibrium assumption was higher than with the highest catch as fished equilibrium assumption (Figure D.4). This happened because this low equilibrium catch level estimated low initial depletion (Figure D.5). The estimates of R_0 increased faster at higher fished equilibrium assumption than the rest of the key estimates (Figure D.6) which were more invariant to alternative equilibrium catch levels. When the minimum required data were used, the accuracy of the key estimates varied more irregularly across alternative fished equilibrium assumptions than when additional data were used (Figure D.5, Figure D.6). Overall, the accuracy of the estimates was higher in a wider range of alternative fished equilibrium assumption when a longer length of data time-series was used than when a shorter data time-series was used (Figure D.6).

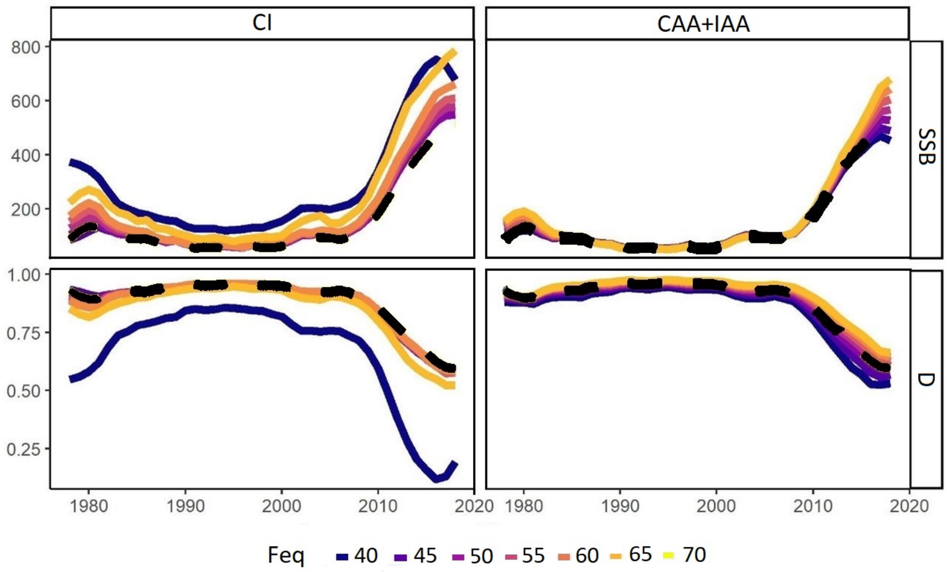


Figure D.4: Estimates of spawning stock biomass (SSB, in 1000 tonnes) and depletion (D) across a range of alternative fished equilibrium assumptions (Feq, in 1000 tonnes) when 41 years data time-series length of total catch and biomass index (CI scenario) and CI and catch-at-age and index-at-age data (CAA+IAA scenario) were used with the $OM_{F_{SS3}}$ population. Each continuous line indicates the estimates of each fished equilibrium assumption, and the black dashed lines indicate the median value of SSB and depletion of the "true" population replicates.

Based on the information available in each data type scenario and the length of the data, the optimum fished equilibrium assumption to obtain the most accurate estimates was different (Figure D.5, Figure D.6). Also, the most accurate values of each key estimate (the lowest MARE values) were found at different fished equilibrium assumptions (Figure D.5, Figure D.6). The initial catch assumption affected the estimates of SSB in the first and last 7 years of the time-series, while the estimates of SSB for the middle of the time-series were robust to the assumed catch as fished equilibrium assumption and close to the "true" population SSB values when the most complete data scenario was used (Figure D.4).

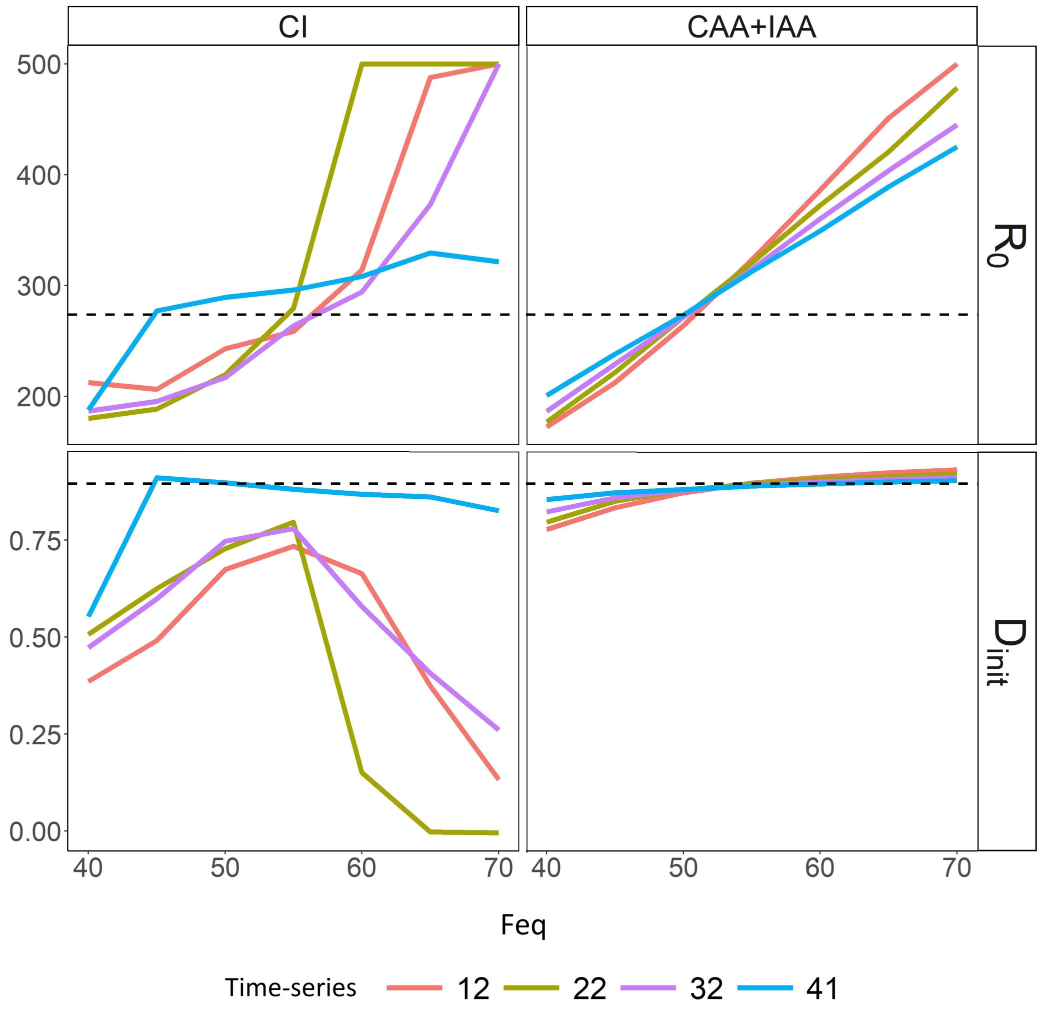


Figure D.5: The median estimates (in rows) of unfished recruitment (R_0 in millions) and initial depletion (D_{init}) when alternative fished equilibrium assumptions (Feq, in 1000 tonnes) (in x-axis), alternative data time-series length scenarios, and the less (CI scenario) and the most (CAA+IAA scenario) complete data type scenarios (in columns) were used with the $OM_{F_{SS3}}$ population. Data type scenarios: catch and index data (CI), and CI and catch-at-age data and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). The black horizontal dashed lines indicate the median of the "true" population R_0 and D_{init}.

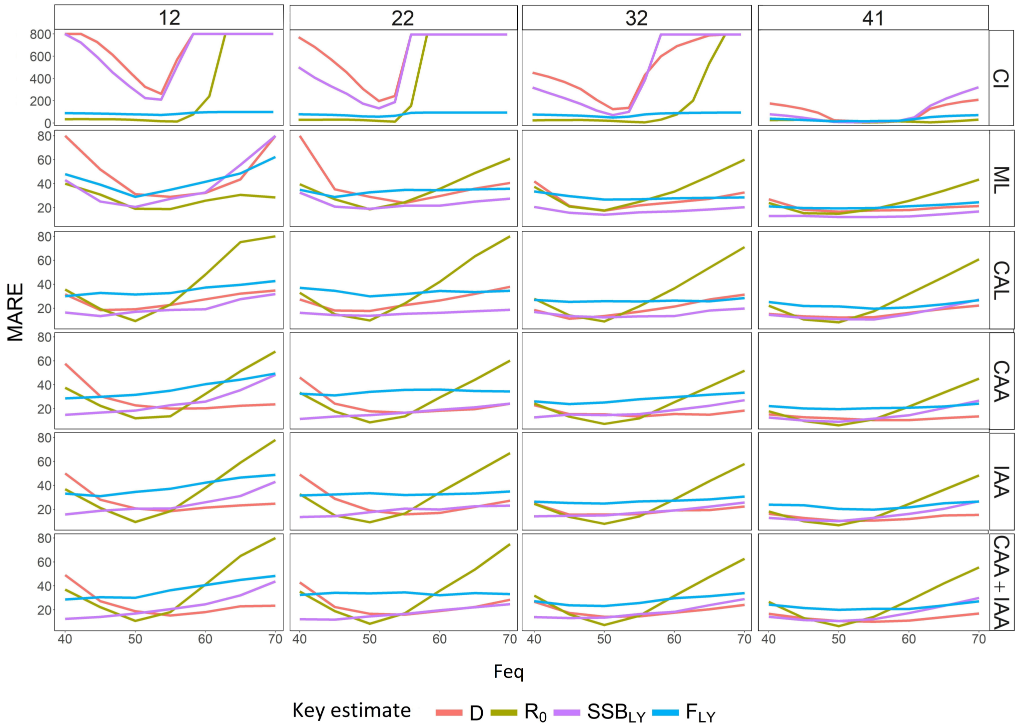


Figure D.6: The median absolute relative error (MARE) of the key estimates across a range of alternative fished equilibrium assumptions (Feq, in 1000 tonnes) for each data type (in rows) and each data time-series length (in columns) scenario combination when the $OM_{F_{SS3}}$ population was used. Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). The y-axis of CI data type plots goes from 0 to 800, whereas the y-axis of the rest of the data type plots goes from 0 to 80. In CI scenario, the MARE values higher than 800 were fixed to 800.

D.3.2.2 Sensitivity analysis of the key estimates to alternative fishery selectivity settings

As expected, the most accurate key estimates were obtained when fishery selectivity was fixed to the “true” population selectivity values (True scenario) (Figure D.7). When catch-at-age data together with index-at-age data were used (CAA+IAA scenario), the accuracy of the key estimates was similar between alternative selectivity scenarios (Figure D.7). However, when catch-at-age or catch-at-length data were used to estimate the selectivity without index-at-age data (CAA and CAL scenarios), the selectivity of the oldest individuals was underestimated (Figure D.8) and the population biomass was overestimated (Figure E.7 in Appendix E.5)). In these scenarios, the accuracy of depletion and SSB_{LY} estimates were low with respect to the rest of the selectivity scenarios (Figure D.7). The accuracy of the key estimates was higher when the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen} scenario) or when the selectivity asymptotic length was fixed to the logistic shape (Logistic scenario) than when this selectivity parameter was estimated (Estimated scenario). The accuracy of the key estimates was similar in both scenarios where the descending limb of the selectivity curve was fixed (Figure D.7).

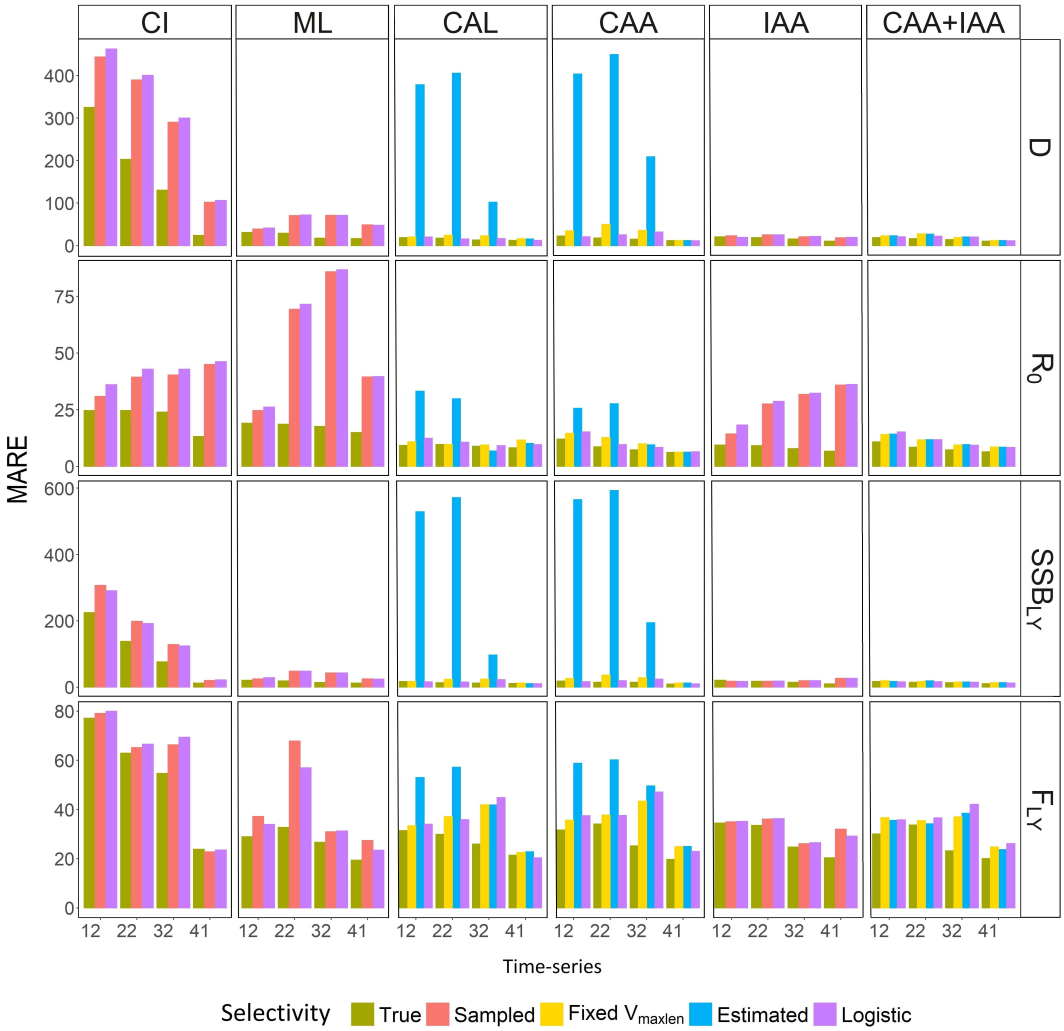


Figure D.7: The median absolute relative error (MARE) of the key estimates (in rows) in each of the five alternative fishery selectivity scenarios for each data type (in columns) and data time-series length (x axis) scenario combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Selectivity scenarios: yearly selectivity was fixed to the "true" values (True), the selectivity parameters were sampled from a range of values uniformly (Sampled), the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen}), the selectivity parameters were estimated (Estimated), and the logistic selectivity was assumed (Logistic). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

The selectivity curves used were different when the selectivity was sampled from the range of values specified in Table D.2 (Sampled scenario) and when the shape of the descending limbs was assumed to be logistic (Logistic scenarios) (Figure D.8). However, the accuracy of the key estimates was similar in both scenarios (Figure D.7).

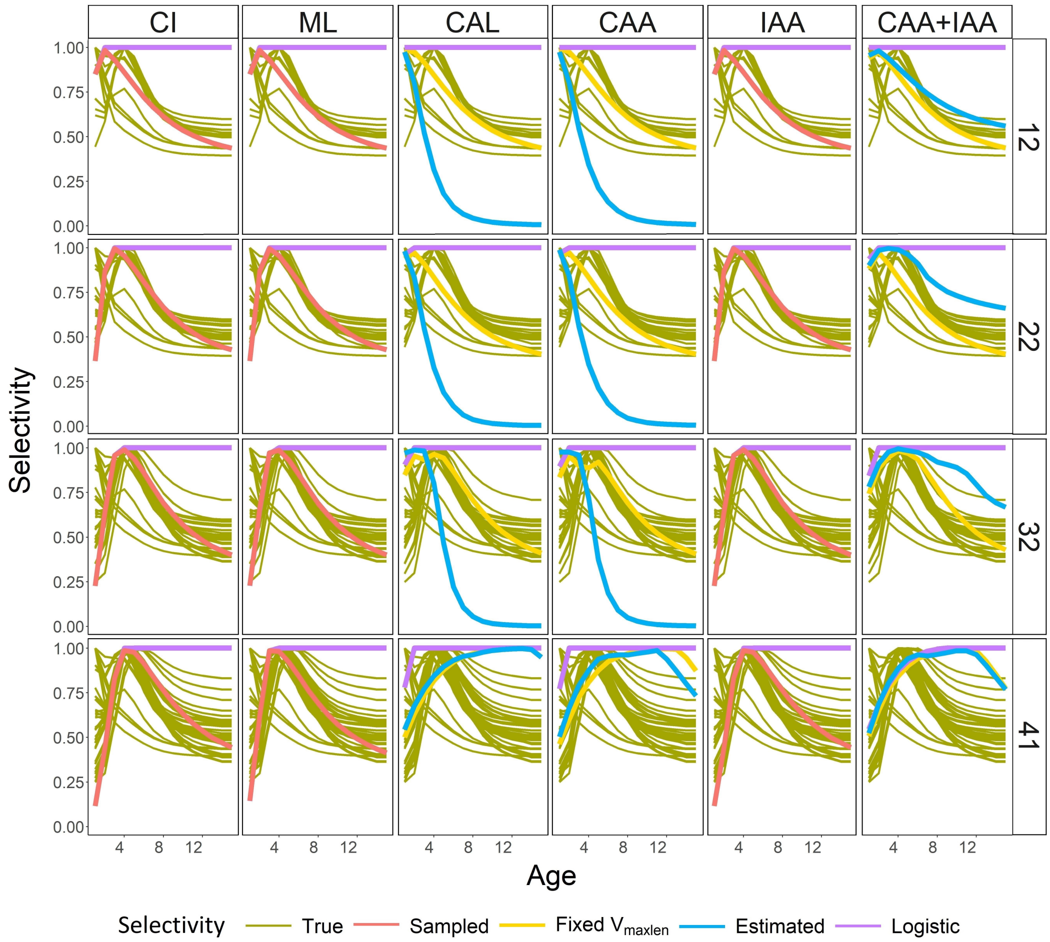


Figure D.8: Selectivity-at-age in each of the five alternative fishery selectivity scenarios for each data type (in columns) and data time-series length (in rows) scenario combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Selectivity scenarios: yearly selectivity was fixed to the "true" values (True), the selectivity parameters were sampled from a range of values uniformely (Sampled), the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen}), the selectivity parameters were estimated (Estimated), and the logistic selectivity was assumed (Logistic). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). The "true" fishery selectivity varied annually, so several lines (one line per year) were plotted. In the rest of the scenarios, the median fishery selectivity of the simulations was plotted because the SRA estimates a constant annual fishing selectivity.

D.4 Discussion

This chapter evaluated by "self-test" simulation the SRA model's performance using alternative lengths and types of datasets, population exploitation level, initial fished equilibrium assumption and misspecification in fishery selectivity. The estimates of the stock's population and exploitation level parameters improved overall in accuracy when more data were used. Furthermore, the longer the data time-series used in the assessment model the higher the accuracy of the estimates.

If the data started from an exploited condition, without any additional data, the length of total catch and biomass index data time-series needs to cover more than two generation times of the stock to obtain accurate estimates. With shorter data time-series, most of the simulations gave estimates that were far from the "true" value and additional data sources were needed to obtain accurate estimates. If the population and data started at unfished conditions, the accuracy of the model estimates was high, except with the shortest data time-series. This could be related to the shape of the fishing mortality trajectory used to generate the unfished populations. Indeed, the shortest data time-series length only covered the start of exploitation and subsequent population decline situation, whereas the rest of the data time-series length scenarios covered the decline and the stabilization of the population biomass. Ono et al. (2014) found that when the data time-series length covered both the population decline and the population stabilization period, the accuracy of the assessment model estimates was higher compared to when the data time-series length only covered the population decline period. The data time-series length of all the scenarios covered the stock population decline period rather than a stable stock population period. Thus, all the data time-series length scenarios analysed in this study covered a relatively informative fishing mortality trajectory according to Ono et al. (2014) and a more constant fishing mortality trajectory could give less accurate estimates than the ones observed in this chapter.

The high accuracy of the estimates observed in this study, may be related to the precision of the biological parameters, where they were fixed at the "true" values in the population when the SRA model was applied. Some biological parameters have a larger impact on model results if specified correctly or incorrectly. Hordyk et al. (2019) demonstrate that uncertainty in maturity parameter would be relatively unimportant compared to uncertainties in other parameters or data sources. However, they also found that bias in natural mortality rate would result in meaningful differences with respect to the equivalent bias in catch and index data, regarding the estimates of long-term yields and stock biomass. When the stock biological parameters are unknown, the accuracy of the assessment model estimates would thus be lower than the accuracy observed in this chapter.

Stock assessment models can use different datasets to estimate stock biomass and exploitation levels (Maunder and Punt, 2013); the collecting cost of each dataset can vary substantially (Begg et al., 2005; Dennis et al., 2015). Simpler and less expensive data collection programs could be sufficient to obtain adequate data on the mean length in catch, rather than that required for good catch-at-length or catch-at-age data. The cost of collecting age-structured data is also higher in terms of personnel needed for collection and age reading capacity compared to the cost of collecting length-structured data. Additionally, fishery data are less expensive than fishery-independent data to collect. However, the fishery data may not give enough information about population abundance, and fishery-independent data could be critical to obtain accurate estimates of the population, as shown in this study when selectivity was estimated from fishery data. Having fishery-independent data can allow you to better understand population dynamics. Data-limited stocks are usually allocated a small amount of research time and investment for data collection and analysis (Bentley, 2014), so fishery-independent data are not available. Ono et al. (2014) demonstrated that the use of additional data in an assessment model is more important for short-lived, fast-growing stocks. In this study, a medium-lived, medium-growing stock was used (ICES, 2019a); the importance of each data type and time-series length to obtain accurate estimates derived from this study may not therefore be applicable to stocks with different life-history such as longevity and growth.

The inclusion of mean length catch data in the SRA model increased the estimates' accuracy less than either length-structured or age-structured data were included. This was specially the case for estimating accurate unfished recruitment from an assessment model rather than the rest of the estimates. In this study, the age-structured data provided little additional information relative to the length-structured data. Ono et al. (2014) demonstrated that if the coefficient of variation of the length-age relationship is low, the length-structured data allow the tracking of cohorts without the need of age-structured data. The variability in somatic growth of hake was moderately low (CV equal to 0.15) with respect to a survey of values from Then et al. (2015), which would explain the small differences in the accuracy of the key estimates between catch-at-age and catch-at-length data. In general, age-structured data are expected to provide more accurate information about somatic growth, natural mortality and stock-recruitment dynamics than length-structured data (Chen et al., 2003; Magnusson and Hilborn, 2007). However, in this study, these biological parameters were known without error. In a real-world situation, where the biological parameters are not precisely known, the use of age-structured data would result in meaningful differences with respect to the use of length-structured data regarding the estimates' accuracy. Finally, our results confirmed that the accuracy of the key estimates was higher when both catch-at-age and index-at-age data were used together than when they were used separately, as previously observed by Chen et al. (2003) and Ono et al. (2014). This was particularly the case for obtaining accurate estimates when the selectivity was estimated from fishery data.

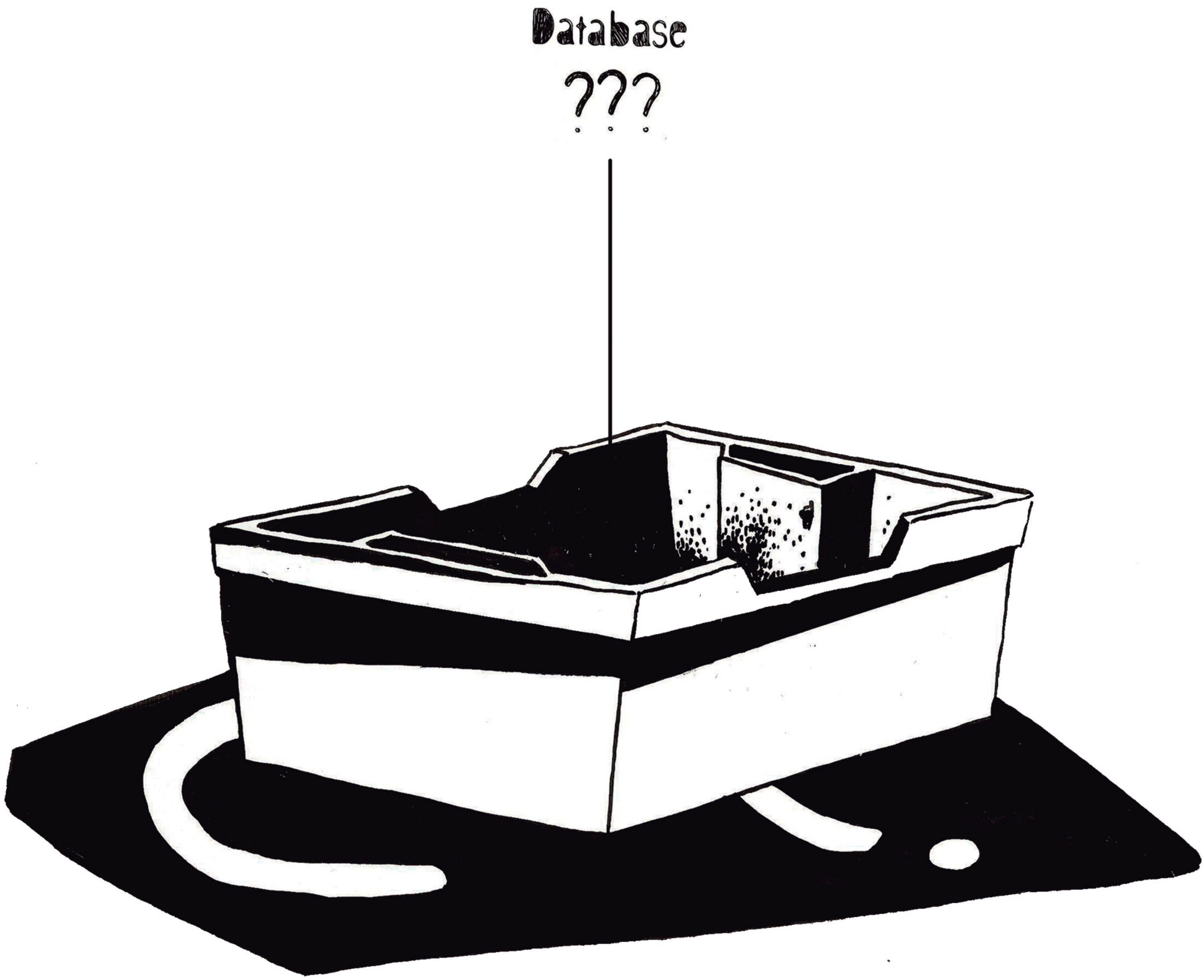
The "true" fishery selectivity pattern showed high variability over time, as would presumably occur in most of the fisheries (Sampson and Scott, 2012). However, this study revealed that assuming time-invariant fishery selectivity, the estimates could be as accurate as those when the fishery selectivity was assumed to be known and time-varying. This study also highlights the need for fishery-independent age-structured data together with fishery age-structured data in an assessment model to obtain accurate estimates when selectivity was estimated from fishery age-structured data. Without index-at-age data, the selectivity estimated from the fishery age- or length-structured data overestimates the population's abundance. The catch of hake is formed mainly by individuals under 7 years old. However, the plus group age for hake is set at 15 years (ICES, 2019a). The selectivity links the population's size and fishery catch data. Without index-at-age data, the available data thus does not provide information about the abundance of the oldest individuals. Hence, when the selectivity was estimated from age- or length-structured catch data, the model assumed that individuals that were not caught by the fishery, only suffered a natural mortality and population abundance was strongly overestimated. Without index-at-age data, if the fishery is focused on small individuals and the abundance of old individuals is low, the results suggested that the descending limit of the selectivity curve must be fixed at any value higher than 0.25 to obtain accurate estimates from an assessment model. In this case, a logistic selectivity curve was the most appropriate selectivity assumption to obtain accurate estimates. However, when old individuals are not really available to the fishery, as occurs in some elasmobranch fisheries (ICES, 2018d), a logistic selectivity assumption may not be appropriate and a lower descending limit could provide more accurate estimates than the logistic ones (ICES, 2018d). Dome-shaped selectivity is observed across a range of species, the logistic selectivity curve is nevertheless the most precautionary selectivity assumption, because it assumes that upwards of some ages all individuals are fully vulnerable to the fishing activity, which produces lower estimates of biomass. Hordyk et al. (2019) analysed the risk of wrong management advice associated to bias in selectivity of the oldest individual fish. In all the selectivity misspecification scenarios, they fix the descending limit of the selectivity curve at alternative values, always higher than 0.25, demonstrating that misspecification in selectivity of the asymptotic length is relatively unimportant compared to other uncertainties (e.g., bias in natural mortality, steepness, or catch or index data).

The impact of alternative fished equilibrium assumption on the results was high when only catch and index data were used and low with additional data, as additional data provided information about initial fishing mortality rate and initial population. The estimates of unfished recruitment, SSB and depletion increased when higher equilibrium catches were assumed. The higher the assumed equilibrium catch level, the higher the population biomass needed to support this catches in a long-term maintaining a constant population biomass (i.e., the higher the unfished recruitment and SSB in the first year), and the higher the depletion of the population in the first year of the simulation. The estimates of depletion were less impacted by the fished equilibrium assumption than the estimates of unfished recruitment and unfished population biomass. The impact of the fished equilibrium assumption was low for the estimates in the middle of the data time-series years. The fished equilibrium assumption mainly affects the estimates corresponding to the first and last 7 years of the model. The estimates from the first and the last years together with the estimates in unfished condition (e.g., unfished recruitment) are the ones routinely used to inform management, such as the biomass at which MSY is attained. The accuracy of the R_0 estimate was strongly driven by the initial depletion of the "true" population. The estimates that benefit the most from long data time-series were the last year fishing mortality and spawning stock biomass estimates. However, in all the data time-series length scenarios, the depletion estimates were more accurate than the rest of the estimates. Depletion is a relative estimate which is usually better estimated than population absolute values, as already found by Punt et al. (2002), Yin and Sampson (2004), Magnusson and Hilborn (2007), and Deroba et al. (2014). When the initial depletion level of the "true" population is unknown (and therefore also the fished equilibrium assumption), the results accordingly suggested that a management plan based on relative estimates (depletion) rather than absolute ones (R_0) could be more adequate. The influence of each estimate's accuracy and bias to give advice on fishing opportunities nevertheless requires further investigation within a closed-loop simulation framework before a management advice is adopted (Hordyk et al., 2019). The alternative of using an assumption about the initial population is to reconstruct the catch time-series until the unfished condition (Pauly and Zeller, 2016). Several organizations, such as ICES (ICES, 2021n), Sea Around Us (Pauly et al., 2020) and FAO (FAO, 2020), are endeavouring to reconstruct and collate historical catch data, which will remain an important challenge in upcoming years.

Quantifying the value of knowing each parameter and the value of having a data-source in assessment models is important to identify the best way to improve the accuracy of population and exploitation level estimates, which would in turn result in better management of the stock. Given the influence of the fished equilibrium assumption on accuracy of the estimates, as revealed by this study, further simulations that incorporate simultaneous bias across biological parameters and initial population assumption, together with other multiple assessment assumptions, should be conducted. Also, the use of priors in biological (e.g., steepness, natural mortality) and scaling (i.e., catchability coefficient) parameters should be analysed. Such future work could take advantage of understanding the SRA model's ability to obtain accurate estimates when they are not precisely known.

In sum, the recommendations derived from this chapter are set out below about the potential use of the SRA model to obtain accurate population dynamics estimates in data-limited situations:

overall, the more data and longer time-series used, the higher the accuracy of the estimates. If the available data starts from an unfished condition, a time-series of total catch and abundance index data with a length equal to one stock generation time would be sufficient to obtain accurate population and exploitation level estimates. If the data starts from an exploited condition, the time-series of total catch and abundance index data will need to cover at least two stock generation times to obtain accurate estimates. If the length of the data time-series is shorter than two stock generation times, additional data is needed. Estimates of mean length in the catch that cover a stock's generation time may be sufficient to obtain accurate relative estimates (e.g., depletion). However, for accurate estimates of unfished recruitment, age- or length-structured data that cover a stock's generation time are needed. If the variability in the stock's somatic growth is low and the biological parameters are known, the use of length-structured or age-structured data will give similar estimates. The addition of one generation time of a stock's index-at-age data together with age- or length-structured catch data is the best option to obtain accurate estimates, especially when the fishery selectivity is unknown. When obtaining length or age data is not possible, improving the initial population assumption is the alternative to obtain accurate estimates. If reconstruction of catch data back in time is not possible, additional information on initial fishing mortality rate is needed to approximate the population's initial depletion and consequently the fished equilibrium assumption. A good initial population assumption would be critical to obtain accurate estimates from an assessment model, particularly when only total catch and abundance index data are available. The mean of the first five years' catch could be the best fished equilibrium assumption; however, adding uncertainty in the assumed fished equilibrium would be beneficial in order to obtain more accurate estimates from an assessment model, especially when only total catch and abundance index data are available.



Appendix E

Supplementary material to Chapter 3

This supplementary material comprises the equations used by the SRA model to estimate parameters (Section E.1) and the ones used in the maximum log-likelihood function (Section E.2). These equations were taken from (Huynh et al., 2020a). Furthermore, the convergence rate of each scenario is given in Section E.3. The performance statistics of the key estimates, when each data type, data time-series length, fishery selectivity and fished equilibrium assumptions with alternative populations were used, are in Section E.4 and Section E.5.

E.1 Estimated parameters

The estimated parameters, denoted in this section as x, are fishing mortality, selectivity parameters, unfished recruitment, recruitment deviations and the initial equilibrium fishing mortality. The catchability coefficient is solved analytically in the model. They are unconstrained over all real numbers and then transformed in order to constrain the corresponding model parameters. For optimization, the transformation is also designed to reduce the scale of all estimated parameters to within an order of magnitude.

The **fishing mortality** (F_y) is estimated from one parameter x^F which is the estimated F in log-space in the middle of the time-series, and all other years were subsequent deviations represented as x_y^{Fdev}:

$$F_y = \begin{cases} exp(x^F) & y \; is \; midpoint \; of \; the \text{ time-series,} \\ exp(x^F) \; * \; exp(x_y^{Fdev}) & y \; is \; not \; midpoint \; of \; the \text{ time-series.} \end{cases} \tag{E.1}$$

The **selectivity parameters** are estimated from age- or length-structured data, and then, the parameters x^{L_5}, $x^{L_{FS}}$ and $x^{V_{maxlen}}$ are estimated over all real numbers:

$$L_5 = 0.99 * L_{FS} - exp(x^{L_5}), \tag{E.2}$$

$$L_{FS} = 0.99 * L_\infty * logit^{-1}(x^{L_{FS}}), \tag{E.3}$$

$$V_{maxlen} = logit^{-1} * (x^{V_{maxlen}}), \tag{E.4}$$

where, L_∞ is the asymptotic length, L_5 is the length of 5% selectivity, L_{FS} is the length of full selectivity, and V_{maxlen} is the selectivity at L_∞.

The **unfished recruitment** (R_0) is estimated in log-space,

$$R_0 = \frac{1}{z} exp(x^{R_0}), \tag{E.5}$$

where z is an optional rescaler, by default set it equal to the mean historical catch, to reduce the magnitude of the x^{R_0} estimate.

The annual **recruitment deviations** (δ_y) are directly estimated in log space from the model.

The **initial equilibrium fishing mortality** (Feq) is equal to 0 when the population is in unfished conditions in the initial year of the simulation. Alternatively, when the model starts from an exploited population, the model estimates the initial equilibrium fishing mortality from the provided fished equilibrium catch in weight using the yield curve.

To scale the population biomass to biomass index values, the **catchability coefficient** (q) is solved analytically as:

$$q \ = \ exp \left(\frac{\sum_y log \ (I_y^{obs}) \ - \ \sum_y log \left(\sum_a N_{y,a} \ w_a \ \right)}{n} \right), \tag{E.6}$$

where I_y^{obs} is the annual observed biomass index data, $N_{y,a}$ is the estimated annual number of individuals-at-age, w_a is the weight-at-age and n is the number of years with index values. The summation is over those n years.

E.2 Maximum log-likelihood

The equations used by the maximum log-likelihood method are presented in this section.

E.2.1 The objective function

The log-likelihood function LL to be maximized is:

$$LL \ = \ \sum_{i\ =1}^{8} \Lambda_i, \tag{E.7}$$

where Λ_i represents each of the log-likelihood component i (i.e., the recruitment deviation, the fished equilibrium assumption, total catch data, abundance index data, abundance index-at-age data, catch-at-age data, catch-at-length data and mean length in the catch data). All the log-likelihood components are equally weighted.

E.2.2 Log-likelihood components

Let θ_i denote the observed value used in the SRA application, and $\widehat{\theta_i}$ the estimated value from the SRA. In all equations subscript y corresponds to year.

The log-likelihood component of the annual **recruitment deviation** in log space (Λ_1) is:

$$\Lambda_1 \ = \ \sum_y \left(-log(\sigma) \ - \ \frac{{\theta_{1y}}^2}{2\sigma^2} \right), \tag{E.8}$$

where σ is the standard deviation of the recruitment deviates. σ is fixed to the value used in the OM ($\sigma = 0.4$)

The log-likelihood component of the **fished equilibrium assumption** (Λ_2) is:

$$\Lambda_2 \ = \ -log\,(0.01) - \frac{\left[log\,(\theta_{2y}) - log\left(\widehat{\theta_{2y}}\right) \right]^2}{2 \ x \ 0.01^2}, \tag{E.9}$$

Λ_2 is set with a small standard deviation (0.01).

The log-likelihood component of annual **catch** data (Λ_3) is:

$$\Lambda_3 = \sum_y \left(-log(0.01) \ - \ \frac{\left[log(\theta_{3y}) \ - \ log(\widehat{\theta_{3y}}) \right]^2}{2 \ * 0.01^2} \right), \tag{E.10}$$

where the standard deviation is fixed at 0.01, meaning that the predicted catch should match the observed catch.

The log-likelihood component of annual **abundance index** data (Λ_4) is:

$$\Lambda_4 = \sum_y \left(-log(\tau_y) \; - \; \frac{\left[log(\theta_{4y}) \; - \; log(\widehat{\theta}_{4y})\right]^2}{2\tau_y^2} \right), \tag{E.11}$$

where τ_y is the standard deviation of the annual abundance index data set it at the value of 0.25 used to generate the observed abundance index data.

The log-likelihood component of the **abundance index-at-age** data (Λ_5) is:

$$\Lambda_5 \; = \; \sum_y ESS_y^a \sum_a \theta_{5y,a} log(\widehat{\theta}_{5y,a}), \tag{E.12}$$

where ESS_y^a is the annual sample size for the age composition data. ESS_y^a was fixed at 50 as done in Magnusson and Hilborn (2007). The subscript a corresponds to age. The age ranged from age 1 to age 15, where the recruitment happened at age 1 and age 15 was the plus group representing all the individuals of maximum age 15 and older.

The log-likelihood component of the **catch-at-age** data (Λ_6) is:

$$\Lambda_6 \; = \; \sum_y ESS_y^a \sum_a \theta_{6y,a} log(\widehat{\theta}_{6y,a}). \tag{E.13}$$

The log-likelihood component of the **catch-at-length** data (Λ_7) is:

$$\Lambda_7 \; = \; \sum_y ESS_y^l \sum_l \theta_{7y,l} log(\widehat{\theta}_{7y,l}), \tag{E.14}$$

where ESS_y^l is the annual sample size for the length compositions data. ESS_y^l was fixed at 125 as used by default in the stock assessment (ICES, 2019a). The length bin l, ranged from 5.5 to 135.5 in intervals of 2 cm.

The log-likelihood component of the **mean length in the catch** data (Λ_8) is:

$$\Lambda_8 \; = \; \sum_y \left(-log\left(\omega_y\right) - \frac{\left[log\left(\theta_{8y}\right) - log\left(\widehat{\theta}_{8y}\right)\right]^2}{2\omega_y^2} \right), \tag{E.15}$$

where ω_y is the standard deviation of the annual mean length catch data and was set to 0.25 as used in the observation model to generate the data.

E.3 Convergence rate

Table E.1: The convergence rate (CR) from the combination of each population, fished equilibrium assumptions (Feq, from 40,000 to 70,000), data type and data time-series length scenario. Selectivity scenarios: yearly selectivity was fixed to the "true" values (True), the selectivity parameters were sampled from a range of values uniformly (Sampled), the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen}), the selectivity parameters were estimated (Estimated), and the logistic selectivity was assumed (Logistic). Data type scenarios (DT): catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). Data time-series length scenarios (TS): 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018).

				TS			
Pop	**Feq**	**Sel**	**DT**	**12**	**22**	**32**	**41**
$OM_{2F_{MSY}}$	0	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
$OM_{F_{MSY}}$	0	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
$OM_{0.5F_{MSY}}$	0	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
$OM_{F_{SS3}}$	40000	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
	45000		CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100

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Table E.1 – Continued from previous page

				TS			
Pop	**Feq**	**Sel**	**DT**	**12**	**22**	**32**	**41**
$OM_{F_{SS3}}$	50000	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
		Sampled	CI	99.42	100	100	90.91
			ML	92.96	88.54	81.58	17.32
			IAA	97.15	97.62	100	99.89
		Fixed V_{maxlen}	CAL	100	100	100	100
			CAA	100	100	100	100
			CAA+IAA	100	100	100	100
		Estimated	CAL	100	100	100	100
			CAA	100	100	100	100
			CAA+IAA	100	100	100	100
		Logistic	CI	99.19	100	100	90.59
			ML	91.66	92.23	81.68	17.66
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	97.27	98.10	100	99.91
			CAA+IAA	100	100	100	100
	55000	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
	60000		CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100
	65000		CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100

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Table E.1 – Continued from previous page

Pop	Feq	Sel	DT	TS 12	TS 22	TS 32	TS 41
$OM_{F_{SS3}}$	70000	True	CI	100	100	100	100
			ML	100	100	100	100
			CAL	100	100	100	100
			CAA	100	100	100	100
			IAA	100	100	100	100
			CAA+IAA	100	100	100	100

E.4 Unfished condition

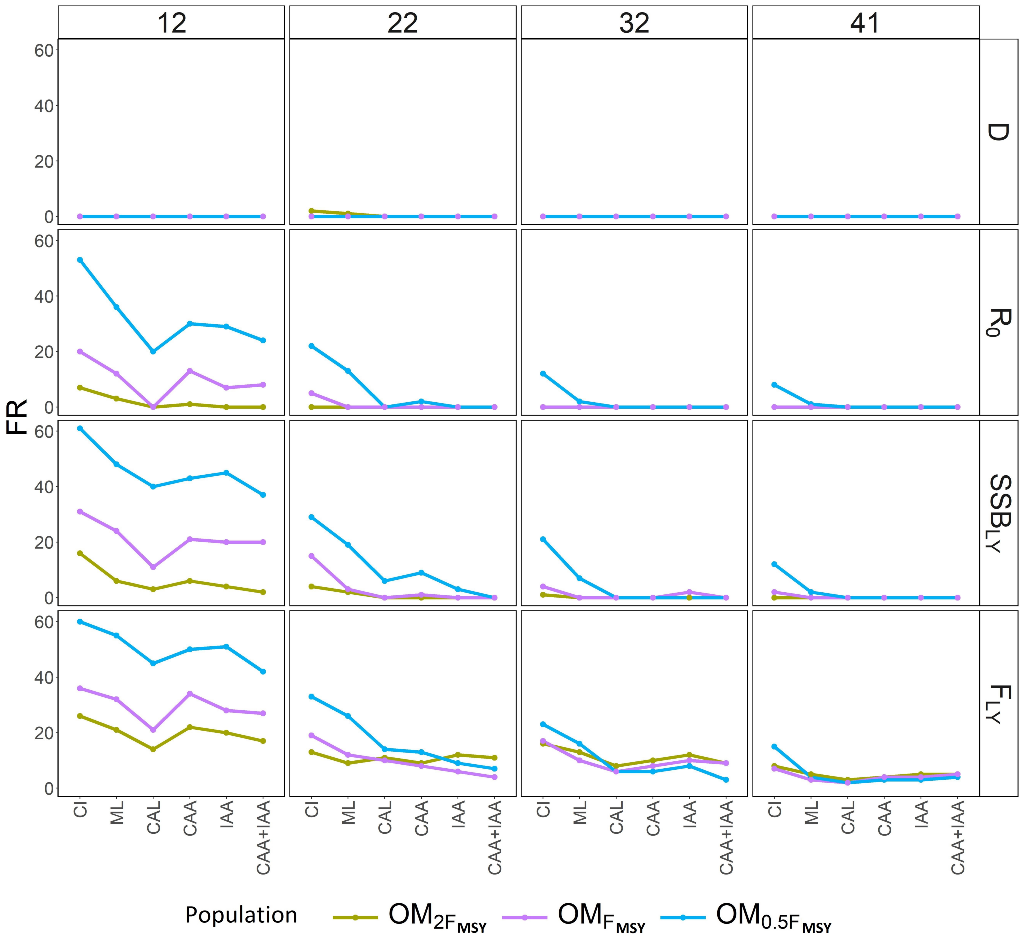

Figure E.1: The failure rate (FR) of the key estimates (in rows) for each data type (x axis) and data time-series length scenario (in columns) combination when the $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ and $OM_{0.5F_{MSY}}$ populations were used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA).

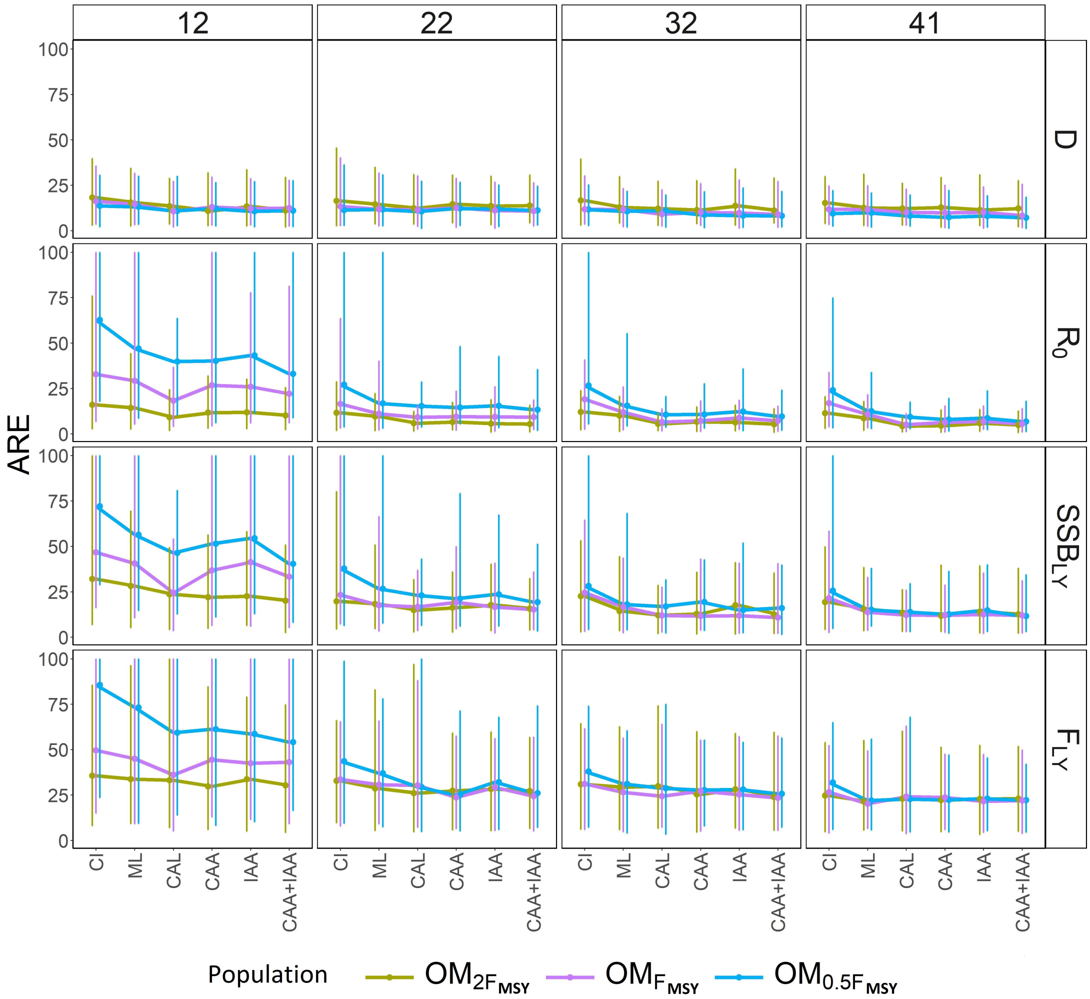

Figure E.2: The 10%, 50% and 90% percentiles of the absolute relative error (ARE) of the key estimates (in rows) for each data type (x axis) and data time-series length scenario (in columns) combination when the $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ and $OM_{0.5F_{MSY}}$ populations were used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). The upper limit of the y-axis was truncated to 100.

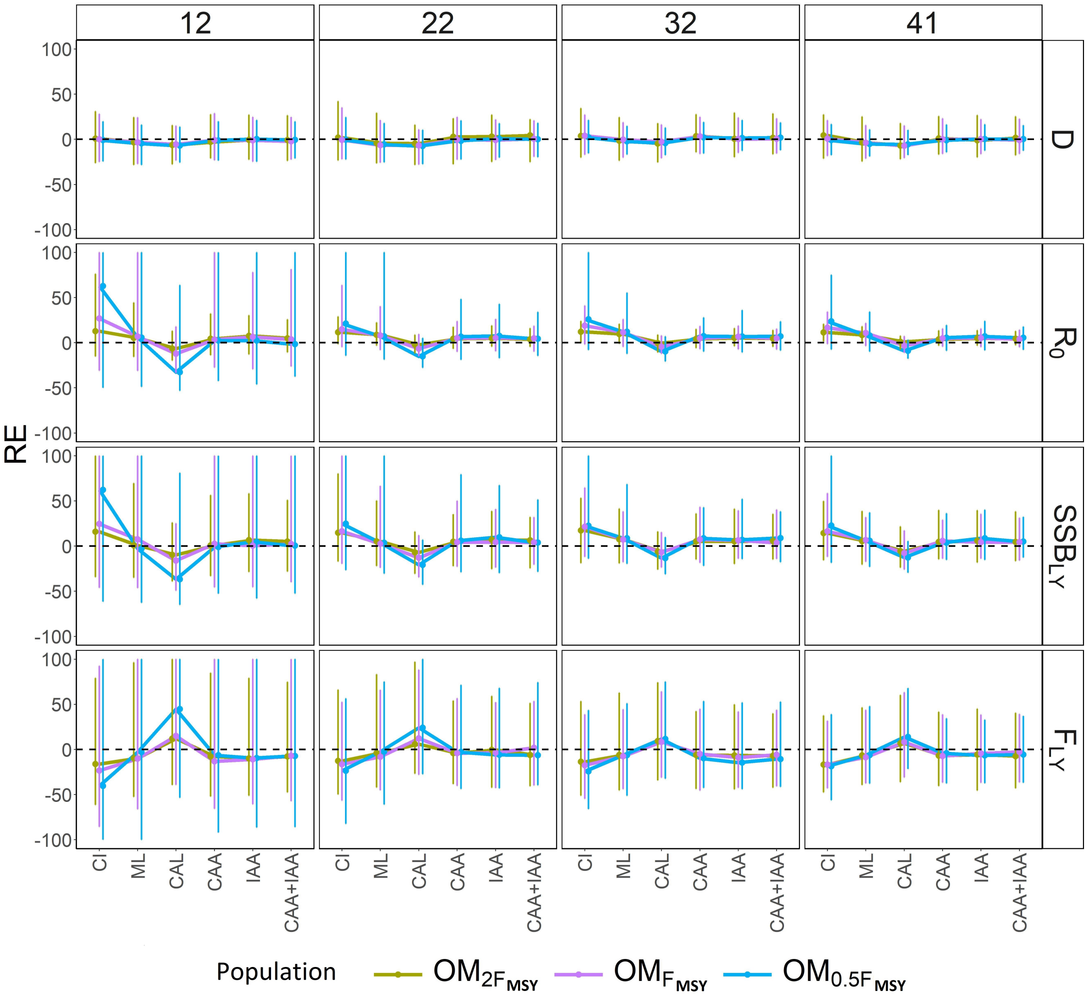

Figure E.3: The 10%, 50% and 90% percentiles of the relative error (RE) of the key estimates (in rows) for each data type (x axis) and data time-series length scenario (in columns) combination when the $OM_{2F_{MSY}}$, $OM_{F_{MSY}}$ and $OM_{0.5F_{MSY}}$ populations were used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_{FL}) and last year fishing mortality (F_{LY}). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). The black horizontal dashed lines indicate no error in the estimates. The upper limit of the y-axis was truncated to 100.

E.5 Exploited condition

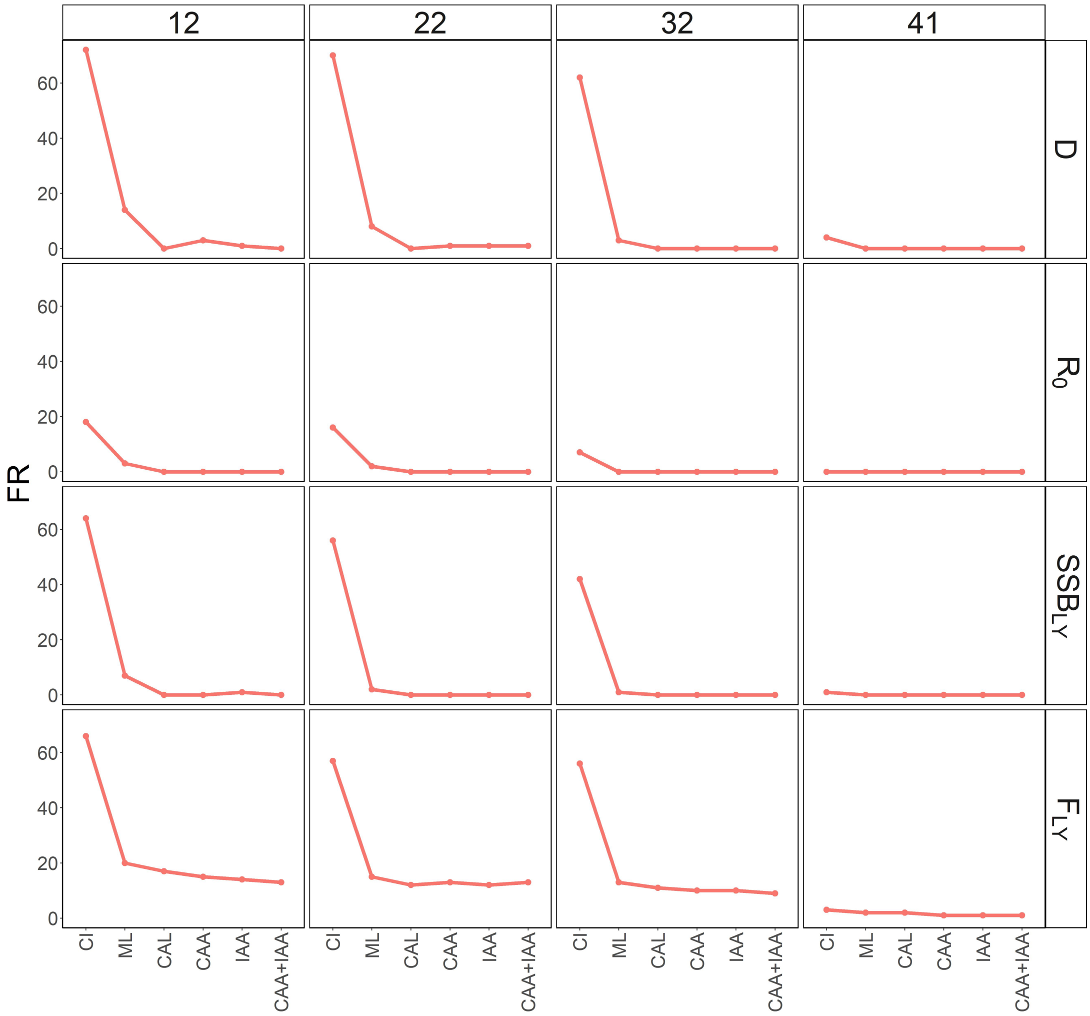

Figure E.4: The failure rate (FR) of the key estimates (in rows) for each data type (x axis) and data time-series length scenario (in columns) combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_FL) and last year fishing mortality (F_LY). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA).

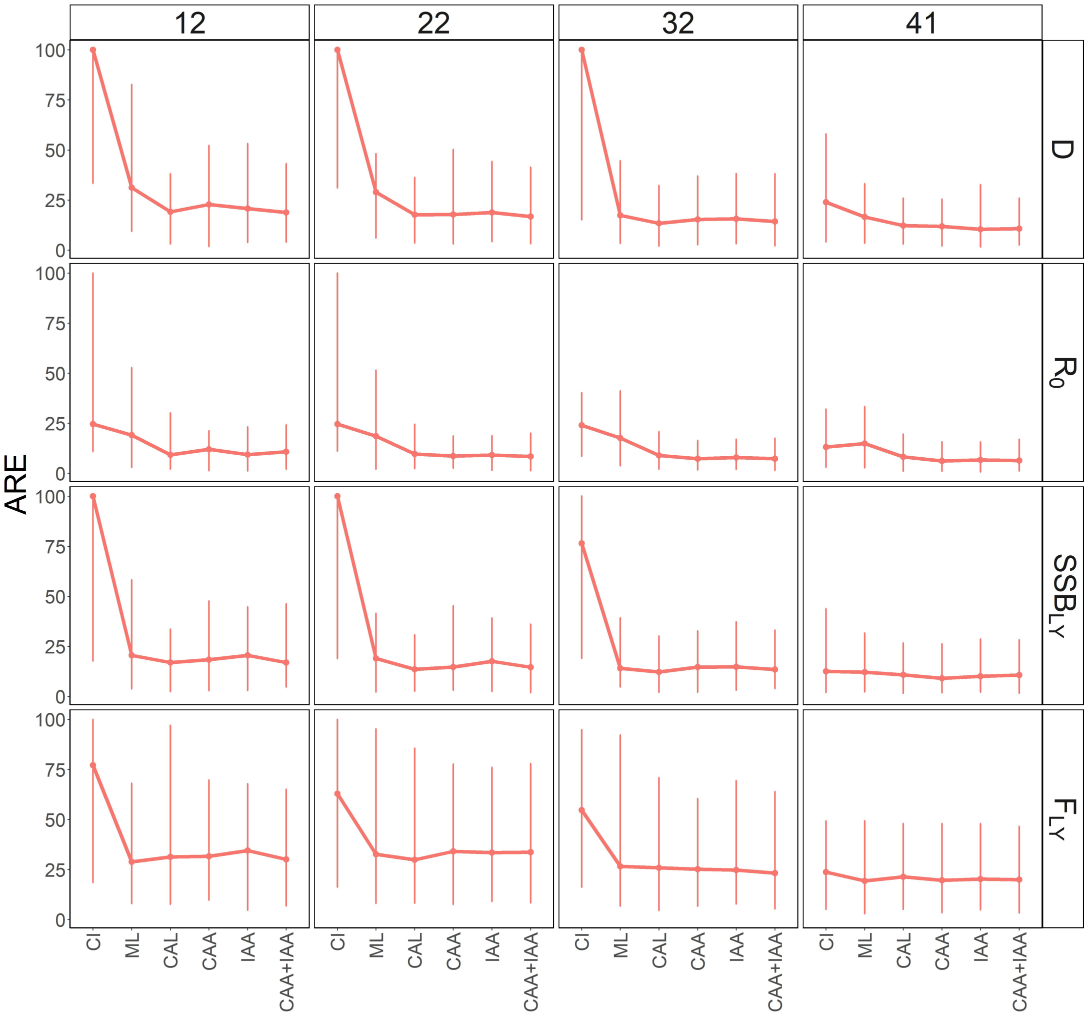

Figure E.5: The 10%, 50% and 90% percentiles of the absolute relative error (ARE) of the key estimates (in rows) for each data type (in columns) and data time-series length scenario (x axis) combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_FL) and last year fishing mortality (F_LY). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). The upper limit of the y-axis was truncated to 100.

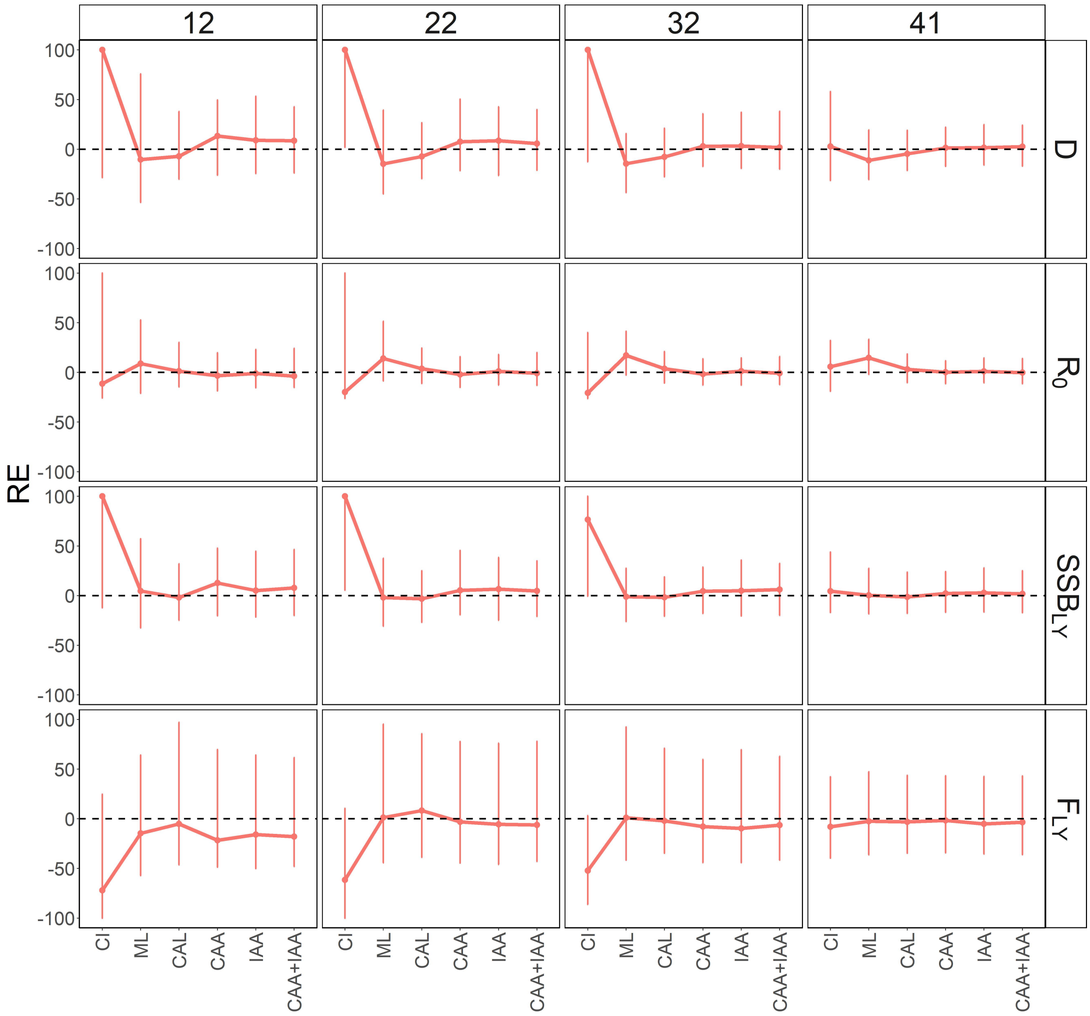

Figure E.6: The 10%, 50% and 90% percentiles of the relative error (RE) of the key estimates (in rows) for each data type (in columns) and data time-series length scenario (x axis) combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_FL) and last year fishing mortality (F_LY). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). The black horizontal dashed lines indicate no error in the estimates. The upper limit of the y-axis was truncated to 100.

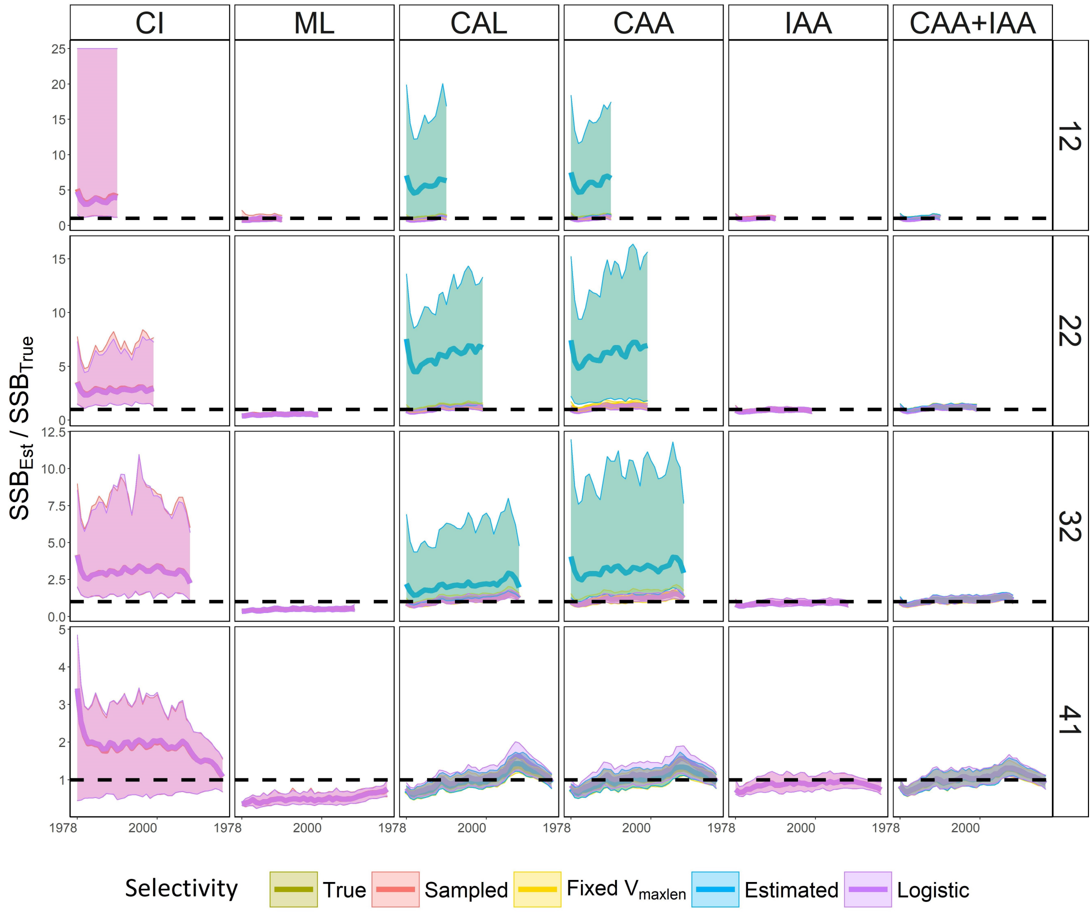

Figure E.7: The estimated spawning stock biomass (SSB_{Est}) relative to the "true" SSB (SSB_{True}) in each of the five alternative fishery selectivity scenarios for each data type (in columns) and each data time-series length (in rows) scenario combination when the $OM_{F_{SS3}}$ population with 50,000 tonnes as fished equilibrium assumption was used. Selectivity scenarios: yearly selectivity was fixed to the "true" values (True), the selectivity parameters were sampled from a range of values uniformely (Sampled), the selectivity at asymptotic length was fixed to the optimum value (Fixed V_{maxlen}), the selectivity parameters were estimated (Estimated), and the logistic selectivity was assumed (Logistic). Data time-series length scenarios: 12 (from 1978 to 1989), 22 (from 1978 to 1999), 32 (from 1978 to 2009) and 41 (from 1978 to 2018). Key estimates: depletion (D), unfished recruitment (R_0), last year spawning stock biomass (SSB_FL) and last year fishing mortality (F_LY). Data type scenarios: catch and index data (CI), CI and mean length catch data (ML), CI and catch-at-length data (CAL), CI and catch-at-age data (CAA), CI and index-at-age data (IAA), and CAA and index-at-age data (CAA+IAA). The upper limit of the y-axis when 12 years data time-series length was used was truncated to 100.

Appendix F

Supplementary material to Chapter 4

This supplementary material describes in detail the conditioning of the stocks to be included in the simulation model to test the performance of current and alternative management strategies of the demersal mixed-fisheries in the Bay of Biscay (Section F.1). In addition, the percentage of total catch in the Bay of Biscay by country are in Section F.2.

F.1 Operating model: Stock dynamics

The closed-loop simulation carried out in Chapter 4 included 28 stocks (Table 4.2). In this section the methodology used to model the dynamics of each of the stocks is described. 13 of the 29 stocks were included with their age-structured population dynamics, and the other 15 stocks were included using a fixed population approach.

F.1.1 Age-structured population

From the 13 stocks included with age-structured population dynamics, nine had an analytical assessment providing abundance and exploitation level estimates. So, these estimates were used to condition their population dynamics in the simulation model. The population dynamics of the other four stocks were conditioned using the SRA model. According to Chapter 3 the data availability of these four stocks (the catch and index data covered less than two generation times but more than one generation time of the stocks and the catch-at-length data was available) was enough to obtain reliable population dynamics estimates from the SRA model.

F.1.1.1 Stocks with quantitative assessment

Hake, black anglerfish, white anglerfish, horse mackerel, mackerel, megrim, sea bass, blue whiting and sole are classified in ICES category 1. Hake and black anglerfish are assessed by a length-structured assessment model, whereas the rest of stocks are assessed by an age-structured models (Table 4.3). In the case of hake and black anglerfish, the assessment outputs were transformed into age-structured data using the probability of being in each length bin given an age class and the mean length-at-age coefficient of variation. Six of these species (i.e., hake, black anglerfish, white anglerfish, horse mackerel, mackerel, megrim) are in the top-ten ranked stocks (Chapter 2).

For all these stocks, the assessments of 2021 were used to condition their population dynamics. As an exception, for black anglerfish the stock assessment outputs of 2022 were used, because until then, it was classified in ICES category 3 and it did not have a quantitative assessment from ICES. The summary of these nine stock assessment models are given in the next nine figures (Figure F.1, Figure F.2, Figure F.3, Figure F.4, Figure F.5, Figure F.6, Figure F.7, Figure F.8, Figure F.9).

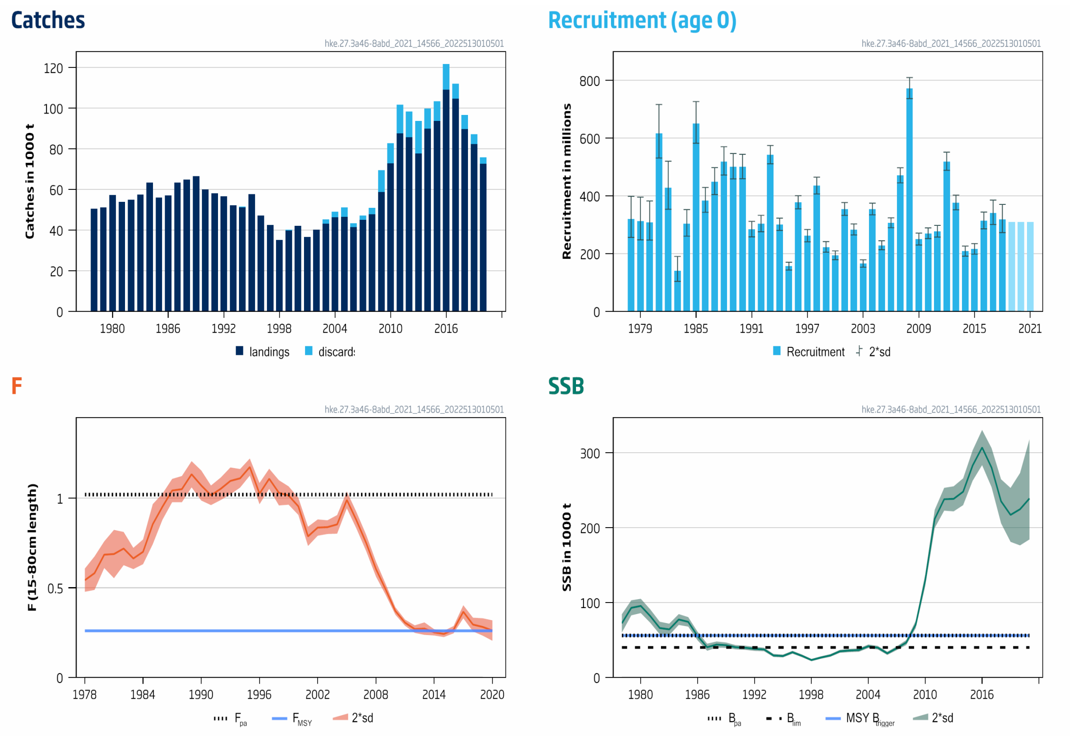

Figure F.1: Hake (*Merluccius merluccius*) in subareas 4, 6, and 7, and divisions 3a, and 8abd, Northern stock. Summary of the stock assessment. Complete discard estimates are available only since 2003. Fishing mortality (F) for length 15-80cm. Recruitment and spawning stock biomass (SSB) plots show 95% confidence intervals (shaded area). Assumed recruitment values are unshaded. Reprinted from ICES (2021c).

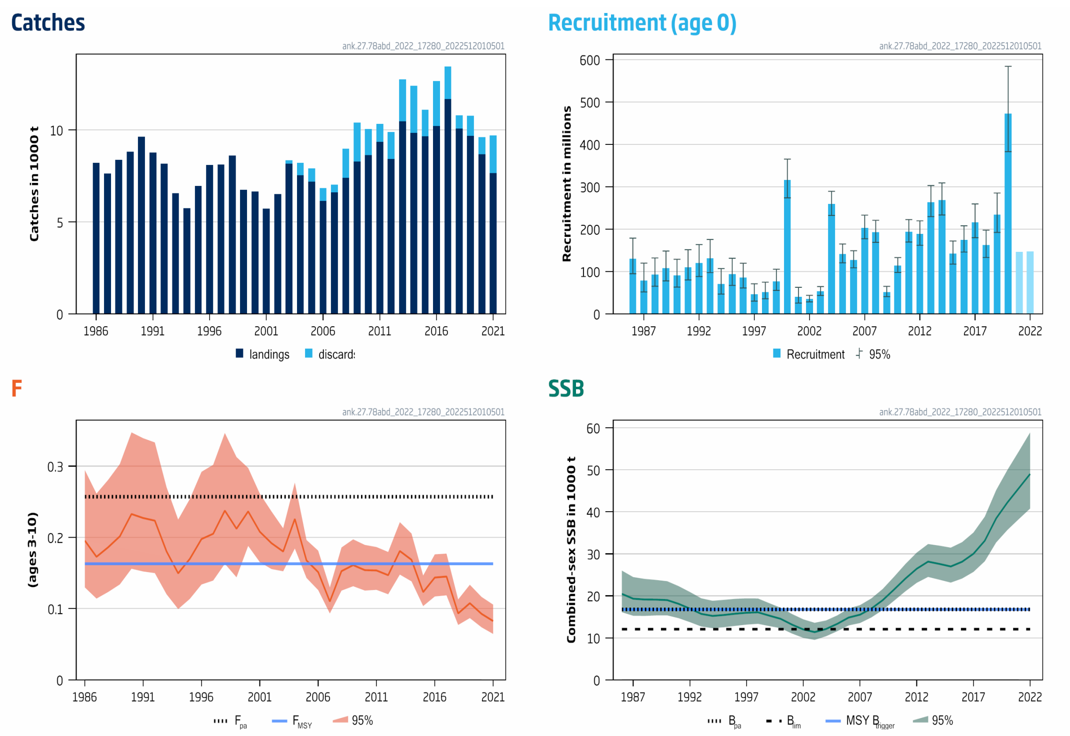

Figure F.2: Black anglerfish (*Lophius budegassa*) in Subarea 7 and divisions 8.a–b and 8.d. Summary of the stock assessment. Discard observations are available since 2003. The assumed recruitment values for 2021 and 2022 are shaded in a lighter color. Reprinted from ICES (2022c).

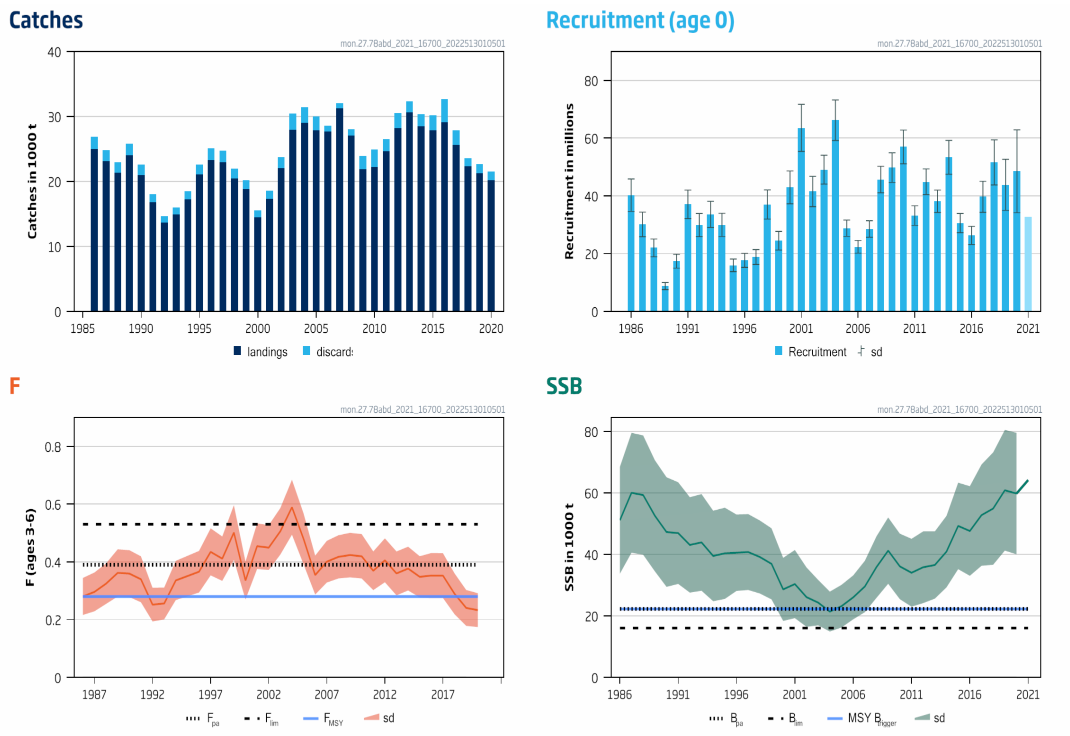

Figure F.3: White anglerfish (*Lophius piscatorius*) in subarea 7 and divisions 8abd. Summary of the stock assessment (weights in thousand tonnes). Discard estimates are available only since 2003. The plots for recruitment, fishing mortality (F), and spawning stock biomass (SSB) show the 95% confidence limits. Assumed recruitment values are unshaded. Reprinted from ICES (2021o).

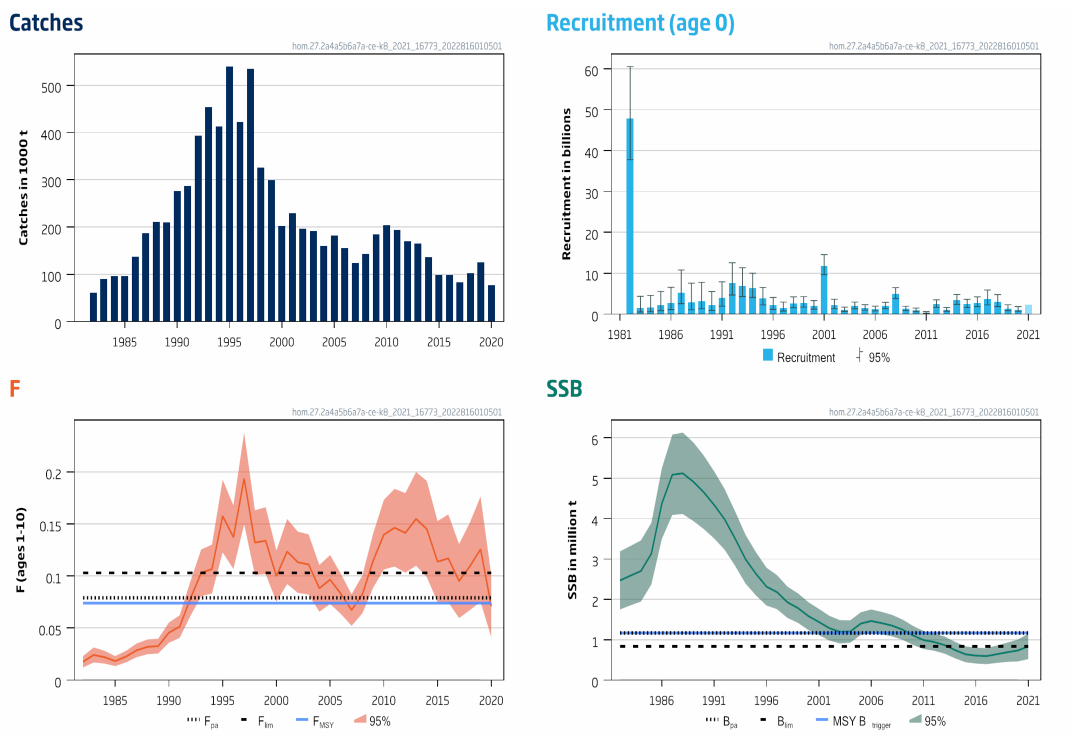

Figure F.4: Horse mackerel (*Trachurus trachurus*) in subarea 8 and divisions 2a, 4a, 5b, 6a, 7a–c, and 7e–k. Summary of the stock assessment. The plots for fishing mortality (F), and spawning stock biomass (SSB) show 95% confidence intervals (shaded area). Assumed recruitment value for 2018 is unshaded. Reprinted from ICES (2021d).

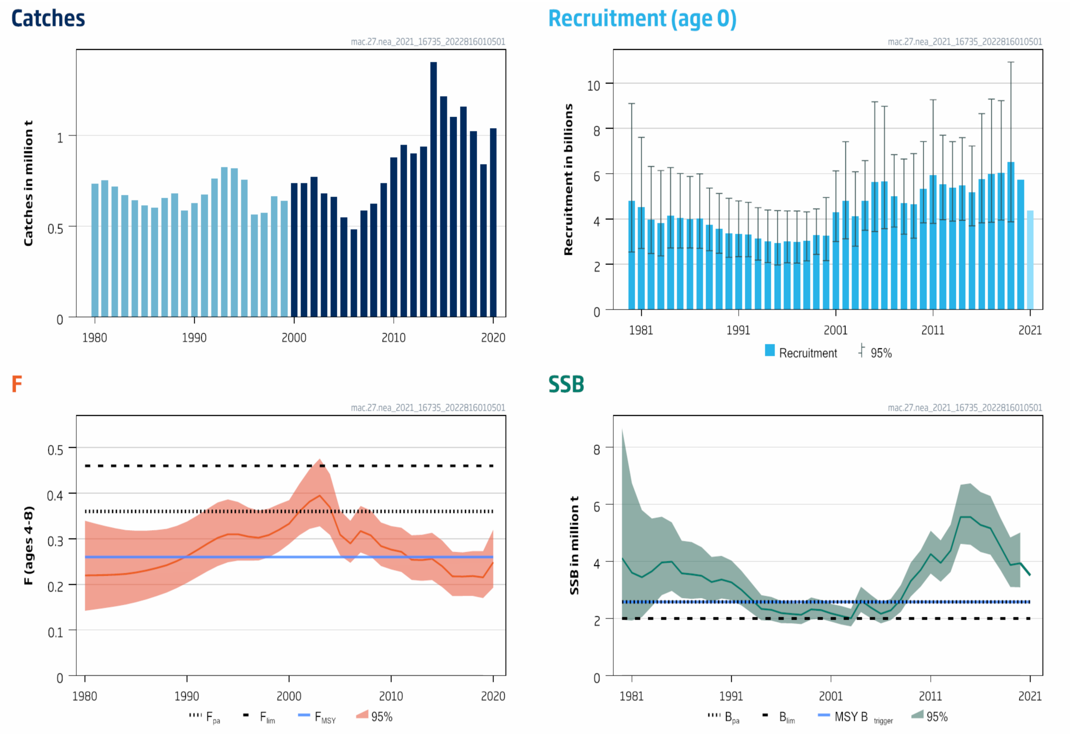

Figure F.5: Mackerel (*Scomber scombrus*) in subareas 1–8 and 14, and division 9a. Summary of the stock assessment. The unshaded catches prior to 2000 are the years that have been down-weighted in the assessment because of the considerable underreporting that is suspected to have taken place. Confidence intervals (95%) are included in the recruitment, fishing mortality (F), and spawning-stock biomass (SSB) plots. Reprinted from ICES (2021f).

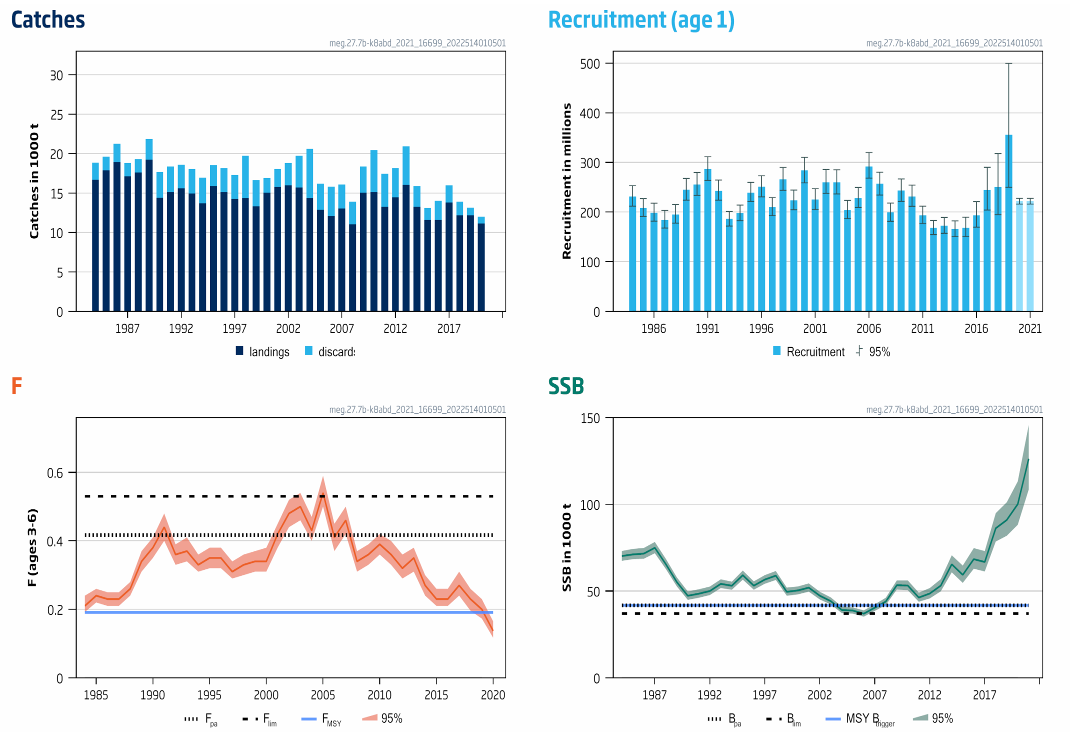

Figure F.6: Megrim (*Lepidorhombus whiffiagonis*) in divisions 7b–k, and 8abd. Summary of the stock assessment. Recruitment, fishing mortality (F), and spawning stock biomass (SSB) are displayed with confidence intervals (95%). Assumed values of recruitment are unshaded.Reprinted from ICES (2021g).

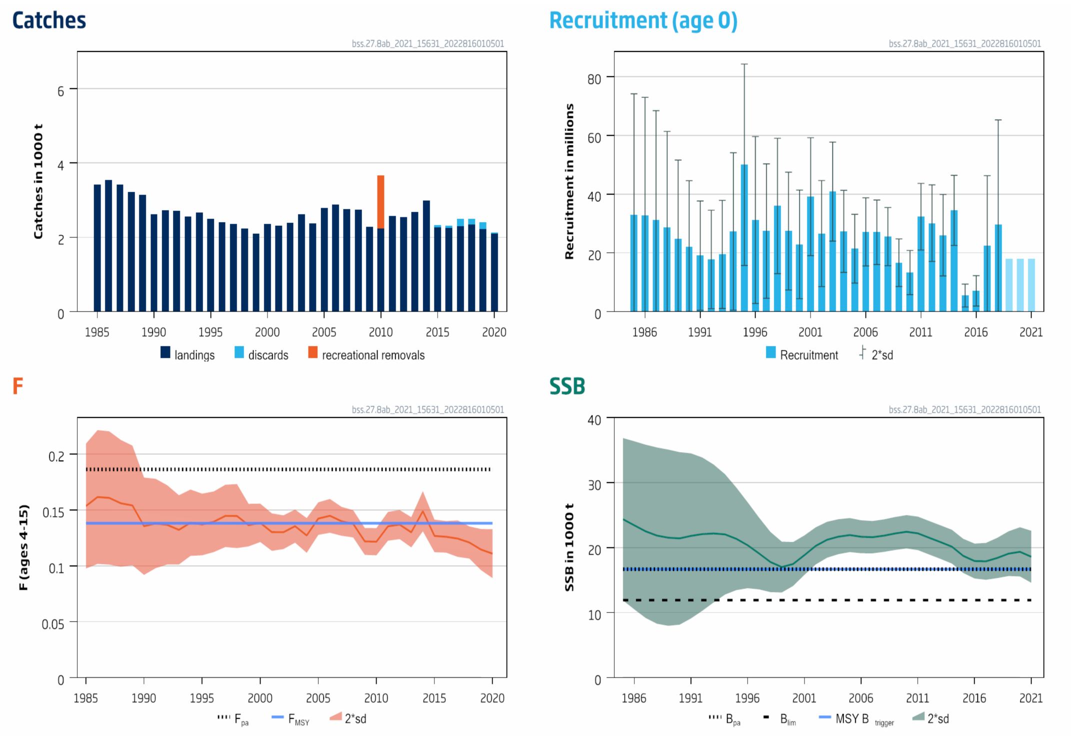

Figure F.7: Sea bass *Dicentrarchus labrax* in divisions 8ab. Summary of the stock assessment (weights in thousand tonnes). Recruitment and spawning stock biomass (SSB) are shown with 95% confidence intervals. Reprinted from ICES (2021k).

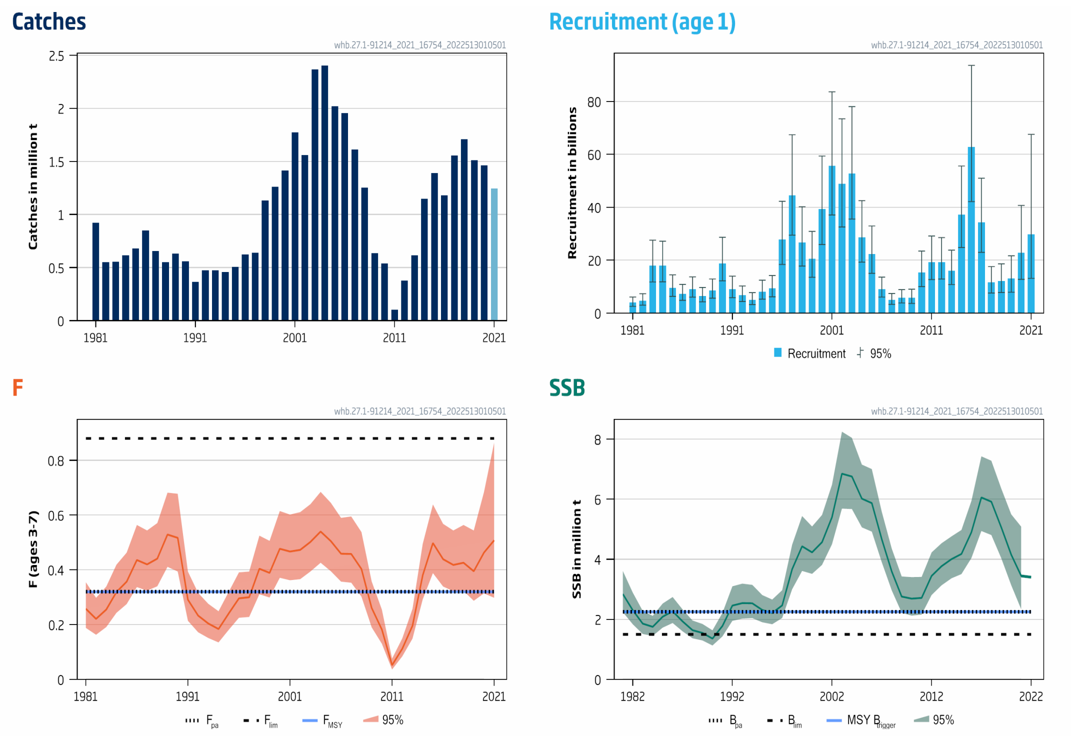

Figure F.8: Blue whiting (*Micromesistius poutassou*) in subareas 1–9, 12, and 14. Summary of the stock assessment. Catches for 2018 (not shaded) are preliminary. For this stock, FMGT = F_{MSY} and SSBMGT= B_{pa}; therefore, the horizontal lines representing these points in the graph would overlap. Confidence intervals (95%) are included in the fishing mortality (F), and spawning-stock biomass (SSB) plots. Reprinted from ICES (2021b).

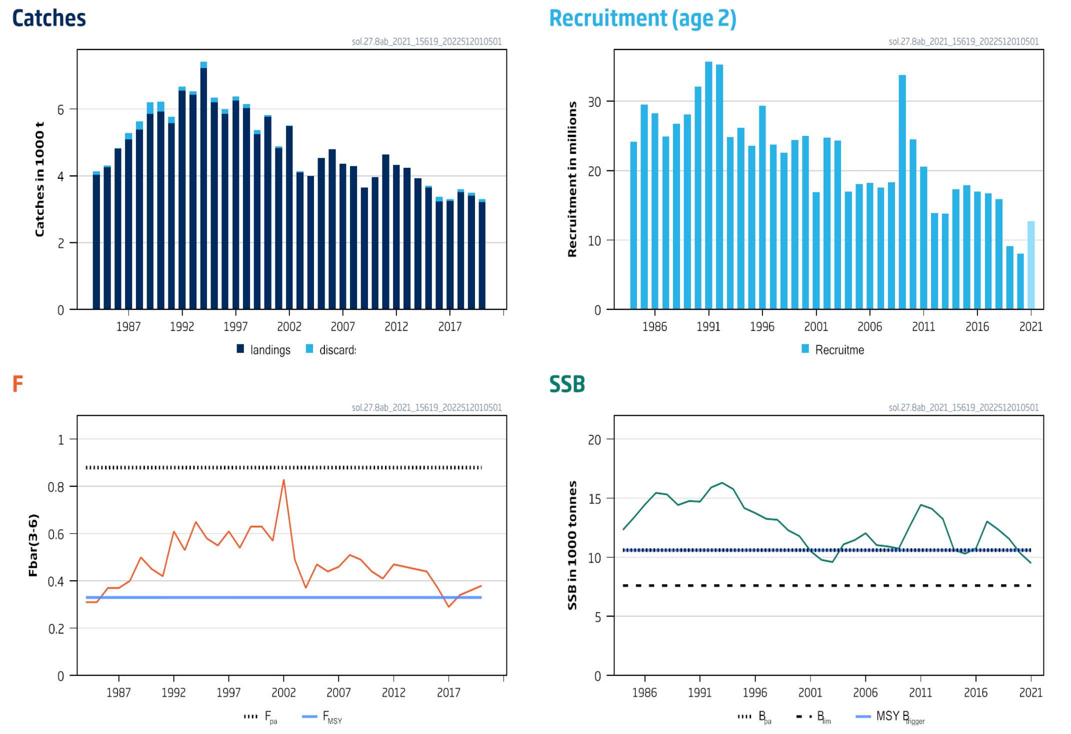

Figure F.9: Sole (*Solea solea*) in divisions 8ab. Summary of the stock assessment. The plots shows assessment outputs catches, recruitment, fishing mortality for ages 3-6 (F_{bar} 3-6), and spawning stock biomass (SSB). Assumed recruitment values are unshaded. The catches plot includes discard estimates only since 2015. Reprinted from ICES (2021m).

F.1.1.2 Stock Reduction Analysis

The stocks classified in ICES category 3 and above do not have a quantitative assessment. However, based on the findings in Chapter 3, the SRA model was used to obtain population estimates for catshark, red mullet, thornback ray and cuckoo ray. For these four stocks, the catch and index data covered less than two generation times but more than one generation time of the stocks, and the catch-at-length data was available, which would be enough to obtain reliable population estimates from the SRA model (Chapter 3). Therefore, the SRA model was applied to condition the population dynamics of these stocks. Specifically, the SRA model available in the MSEtool package version 3.3.0 (Hordyk et al., 2021) within the R environment (R Core Team, 2022) was used.

The main differences between the SRA model available in MSEtool package version 3.3.0 (Hordyk et al., 2021) and the one available in the MSEtool package version 2.0.1 (Huynh et al., 2020a), which was tested in Chapter 3, is that the new version allows for the inclusion of uncertainty in total catch data and the assumption of fished equilibrium, whereas the old version assumed a fixed low error of 0.01. Thus, using the new version of the SRA model, we estimated the population dynamics of catshark, red mullet, thornback ray and cuckoo ray, including uncertainty from all sources (such as, catch data, abundance index data, biological parameters, and initial population assumption).

Catch and Index data The longest available catch, and abundance index data were used (Table F.1). As an exception, the thornback ray and cuckoo ray landings data quality has improved in recent years, where ICES revised elasmobranch landings data for the period 2009-2015 (ICES, 2020a,d) and this revision brought an important increase in historical landing. Thus, for these two stocks the corrected time-series were only used (from 2009 to 2020) in the SRA application. In addition, some corrections of historical landings data were done for red mullet. In fact, around 60% of the total landings of red mullet corresponds to French fleets (ICES, 2020b) and French data are not available for 1999. Thus, we estimated French landings for 1999 with the mean of the closest two years French catches.

Quite complete discards data were only known for catshark from 2011 to 2020 (ICES, 2021e). The survival rate of discards individuals is high (72%) but the recent years discards represent a high percentage with respect to the total catch, being the last eight years discard rate between 80-90% of their total catch (ICES, 2018d). Thus, although the survival rate is high, the dead discards amount is higher than the landings amount and we consider essential to include those dead discards individuals as part of the whole fishing exploitation pattern in catshark assessment. For that, first, discards data from 1997 to 2010 were estimated based on 2011-2013 known landings and discards proportions. Then, using the discards data and the survival rate (72%), the dead amount of individuals in the discards was calculated (ICES, 2018d). This way, in catshark assessment, the sum of the landings and dead discards amount was used. In the rest of the stocks, discards data were unknown or negligible, so only landings data were used (Table F.1). For red mullet discarding is known to take place but are estimated to be negligible (ICES, 2020b). For thornback ray and cuckoo ray discarding is known to take place but has not been fully quantified. The percentage of discarded rays (*Raja spp.*) by the Basque Bottom otter trawl fishery exhibits significant variation, ranging from 0% to 100% by weight, depending on the year (ICES, 2018a). However, the percentage of dead discards falls within a lower range. According to ICES (2018a), the estimated percentage of dead discards for skates in the Bay of Biscay and Iberian Waters is between 4% and 24%.

The French biomass survey index (EVHOE) was used for catshark and red mullet (ICES, 2021e; Ifremer, 2020). For thornback ray and cuckoo ray, the annual mean of two normalized biomass indices was employed (ICES, 2020a,d). Specifically, the biomass index data for thornback ray combined the Spanish (SpGFS) and French (EVHOE) indices, while for cuckoo ray, it incorporated the Irish (IGFS) and French (EVHOE) indices (ICES, 2020a,d).

The catch-at-length data of these four stocks was available only for the Basque demersal mixed-fisheries that operates in the Bay of Biscay and not for all the fleets that catch these stocks. The catches of the Basque demersal mixed-fisheries represent less than 50% of the total catches of each stock. However, in Chapter 3 the importance of length-structured data to obtain accurate estimates from an assessment model was highlighted, so we decided to use them in the SRA application together with catch and biomass index data.

Table F.1: Catch and index time series data used to condition the age-structured population dynamics of catshark, red mullet, thornback ray and cuckoo ray by Stock Reduction Analysis model (SRA). EVHOE = FR-EVHOE-WIBTS-Q4 biomass survey, SBTS = the annual mean of two normalized biomass indices (SpGFS-WIBTS-Q4 and FR-EVHOE-WIBTS-Q4), IBTS= the annual mean of two normalized biomass indices (IE-IGFS-WIBTS-Q4 and FR-EVHOE-WIBTS-Q4), IObs, is the index observation error and CObs is the catch observation error. Dead discards means the dead amount in discards. ^{rec} means reconstructed data.

Species	Data	Type	Setting	References
Catshark	Catch	Landings	1997-2020	ICES (2021e)
		Dead discards	1997-2010^{rec}, 2011-2020	
		Catch-at-length	2005-2020	AZTI fisheries database
		CObs	0.1	
	Index	EVHOE	1997-2020 (except 2017)	ICES (2021e)
		IObs	0.1	
Red mullet	Catch	Landings	1975-2020	ICES (2017, 2020b)
		Catch-at-length	2009-2020	AZTI fisheries database
		CObs	0.1	
	Index	EVHOE	1997-2019	Ifremer (2020)
		IObs	0.1	
Thornback ray	Catch	Landings	2009-2020	ICES (2020d)
		Catch-at-length	2009-2020	AZTI fisheries database
		CObs	0.1	
	Index	SBTS	2009-2019	ICES (2020d)
		IObs	0.1	
Cuckoo ray	Catch	Landings	2009-2020	ICES (2020a)
		Catch-at-length	2009-2020	AZTI fisheries database
		CObs	0.1	
	Index	IBTS	2009-2019	ICES (2020a)
		IObs	0.1	

Parameters We used an age-range of each stock where the recruitment happens at age 0 and the maximum age is the plus group age (Table F.2). Recruitment was modelled based on a Beverton and Holt stock-recruitment relationship. The recruitment was parameterized using steepness (h) and the coefficient of variation of lognormal recruitment deviations (σ). h is unknown for all the stocks analysed. Thus, following the Wiff et al. (2018) recommendation, it was derived from the relationship between the length at which 50 percent of the individuals are mature (L_{50}) and the asymptotic length (L_{∞}). σ of catshark and red mullet were unknown and they were set at 0.4. For thornback ray and cuckoo ray σ was set at 0.4 based on Ormseth and Matta (2011). For all the stocks, growth was modelled according to the von Bertalanffy growth function. Natural mortality (M) was assumed constant across the age groups. When sex-combined data for maturity, natural mortality, and/or growth parameters were unavailable in the literature, and only sex-disaggregated values were accessible, the sex-disaggregated values were combined by assuming an equal 50% sex ratio.

All the stocks biological parameters were taken from literature (Table F.2). When it was possible, the studies done considering individuals from the stock distribution area were used (Table F.2). Catshark life-history parameters were obtained from studies conducted in the Cantabrian Sea (Rodríguez-Cabello et al., 2005, 1998, 2018), and North Atlantic sea (Heesen et al., 2015). Red mullet life-history parameters were taken from studies conducted in the Bay of Biscay (N'Da, 1992), and English Channel and southern North Sea (Mahé et al., 2013). Thornback ray and cuckoo ray life-history parameters were obtained from studies conducted in Iberian coast (Serra-Pereira et al., 2008, 2011), Celtic Sea (McCully et al., 2012), British Isles (Ellis et al., 2013; Gallagher et al., 2004; McCully et al., 2012) and Alaska (Ormseth and Matta, 2011). For the assessment of these two stocks, the natural mortality calculated for undulate ray in Portuguese continental coast was used (Serra-Pereira et al., 2013).

The information of the historical fishing exploitation trajectory of these stocks is limited, and the catch data could not be extrapolated back in time until the unfished condition using any reconstruction method. Thus, the fished equilibrium condition was assumed at the first year of the time-series as the mean of the first five years catches (Table F.2).

Table F.2: Biological parameters and values used to condition the population dynamics of catshark, red mullet, thornback ray and cuckoo ray by Stock Reduction Analysis. Age range, where $^{+}$ means the plus group age; the initial population was estimated from the catch in equilibrium assumption (Feq), witch is the mean of the first five years catch in tonnes. Recruitment compensation ratio or steepness (h); standard deviation of the recruitment deviates (σ); length-weight relationship parameters (a, and b); von Bertalanffy growth parameters (asymptotic length (L_∞), slope (k), and initial size (t_0)); maturity parameters (length at 50 percent maturity (L_{50}) and length at 95 percent maturity (L_{95})) and natural mortality (M).

Stock	**Process**	**Parameter**	**Setting**	**Reference**
Catshark	Age range		1-15^{+} age	ICES (2018d)
	Initial pop	Feq	1042.6	1997-2001 mean catches
	Recruitment	h	0.78	Wiff et al. (2018)
		σ	0.4	*Unknown*
	Length-weight	a	2.20E-06	Heesen et al. (2015)
		b	3.12	
	Growth	L_∞	69.3cm	Rodríguez-Cabello et al. (2005)
		k	0.21	
		t_0	-0.76	
	Maturity	L_{50}	54.2cm	Rodríguez-Cabello et al. (1998)
		L_{95}	61.2cm	
	Natural mortality	All ages	0.26	Rodríguez-Cabello et al. (2018)
Red mullet	Age range		1-6^{+} age	ICES (2018c)
	Initial pop	Feq	795	1975-1979 mean catches
	Recruitment	h	0.41	Wiff et al. (2018)
		σ	0.4	*Unknown*
	Length-weight	a	3.28E-06	Mahé et al. (2013)
		b	3.24	
	Growth	L_∞	37.73cm	Mahé et al. (2013); N'Da (1992)
		k	0.29	
		t_0	-2.61	
	Maturity	L_{50}	15.5cm	ICES (2018b); Mahé et al. (2013)
		L_{95}	29.5cm	
	Natural mortality	All ages	0.4	ICES (2015a)

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Table F.2 – Continued from previous page

Stock	Process	Parameter	Setting	Reference
Thornback ray	Age range		1-12+ age	ICES (2018a)
	Initial pop	Feq	424.6	2009-2013 mean catches
	Recruitment	h	0.57	Wiff et al. (2018)
		σ	0.4	Ormseth and Matta (2011)
	Length-weight	a	4.50E-06	Ellis et al. (2013)
		b	3.1	
	Growth	L_∞	128cm	Serra-Pereira et al. (2008)
		k	0.112	
		t_0	-0.62	
	Maturity	L_{50}	73cm	Serra-Pereira et al. (2011)
		L_{95}	80cm	
	Natural mortality	All ages	0.27cm	Serra-Pereira et al. (2013)
Cuckoo ray	Age range		1-10+ age	ICES (2018a)
	Initial pop	Feq	3767.2	2009-2013 mean catches
	Recruitment	h	0.71	Wiff et al. (2018)
		σ	0.4	Ormseth and Matta (2011)
	Length-weight	a	3.60E-06	McCully et al. (2012)
		b	3.126	
	Growth	L_∞	79.2cm	Gallagher et al. (2004)
		k	0.24	
		t_0	-0.57	
	Maturity	L_{50}	54.05cm	McCully et al. (2012)
		L_{95}	57.5cm	
	Natural mortality	All ages	0.27	Serra-Pereira et al. (2013)

Uncertainty Catshark, thornback ray and cuckoo ray are classified in ICES category 3, while red mullet is in ICES category 5. The available data on their indices, catches, and biological parameters are known to be uncertain. However, the specific range of uncertainty for each data and parameter remains unknown. To address this, the uncertainty of all the data and biological parameters in the SRA application was accounted for by utilizing different coefficients of variation within a uniform distribution. This approach aimed to encompass both the real values and the existing uncertainty.

The coefficient of variation of catch and biomass index data was fixed to 10% in a uniform distribution (Table F.1). In the most uncertain biological parameters, which are the natural mortality and the recruitment parameters, the uncertainty was introduced applying a uniform distribution centered at the mean parameter values with a CV of 10% (Table F.2). For more certain parameters however, i.e., in length-weight relationship, growth and maturity parameters, a coefficient of variation of 5% was used (Table F.2). Uncertainty in fished equilibrium assumption was generated by multiplying the mean of the first five years catches with a uniform error sampled with mean equal to 1 and a coefficient of variation equal to 0.25.

Selectivity of the fleet was defined based on the length frequency distribution data used in the SRA application. However, as recommended in Chapter 3, the minimum and maximum of the descending limit of the selectivity curve were fixed. For elasmobranch old individuals are not really available to the fishery so a logistic selectivity assumption may not be appropriate and a lower descending limit could provide more accurate estimates than the logistic ones (ICES, 2018d). Thus, for catshark, thornback ray and cuckoo ray, the descending limit of the selectivity curve was sampled from 0.25 to 0.8 in a uniform distribution when the SRA model was applied. However, for red mullet, as in the northern stock assessment of this species, logistic selectivity pattern was assumed (ICES, 2018b).

Fitting 100 model fits were carried out creating 100 stock trajectories that matched the observed catch and index values given the biological and selectivity parameters. From these 100 population dynamics trajectories, those with initial population higher than 5% of the virgin biomass were considered as plausible populations dynamics. Sampling randomly 1000 times from these 100 plausible population dynamics, we finally obtained the 1000 population dynamics trajectories to be included in the simulation framework in Chapter 4.

The 1000 plausible stock trajectories for catshark, red mullet, thornback ray and cuckoo ray are shown in Figure F.10, Figure F.11, Figure F.13 and Figure F.12 respectively.

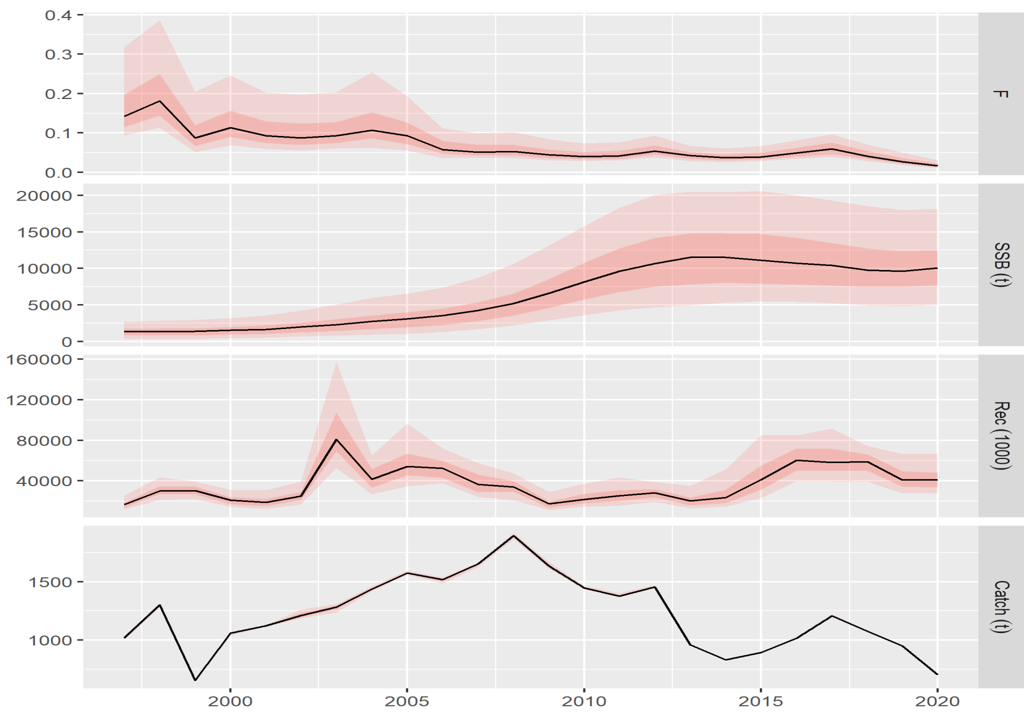

Figure F.10: Assessment summary plots of catshark (*Scyliorhinus canicula*) in divisions 8abd using Stock Reduction Analysis (SRA) approach.Their 5%, 25%, 50%, 75% and 95% quantiles are shown as a black line (median, 50% quantile) and red ribbons.

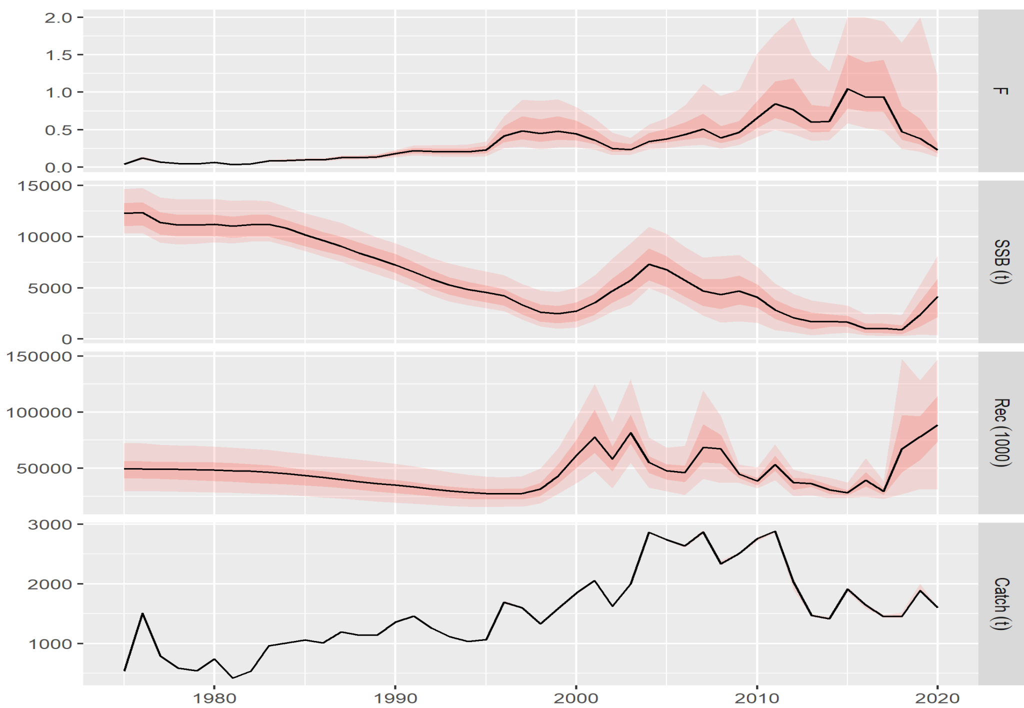

Figure F.11: Assessment summary plots of red mullet (*Mullus surmuletus*) in subareas 6 and 8, and divisions 7a–c, 7e–k, and 9a using Stock Reduction Analysis (SRA) approach. Their 5%, 25%, 50%, 75% and 95% quantiles are shown as a black line (median, 50% quantile) and red ribbons.

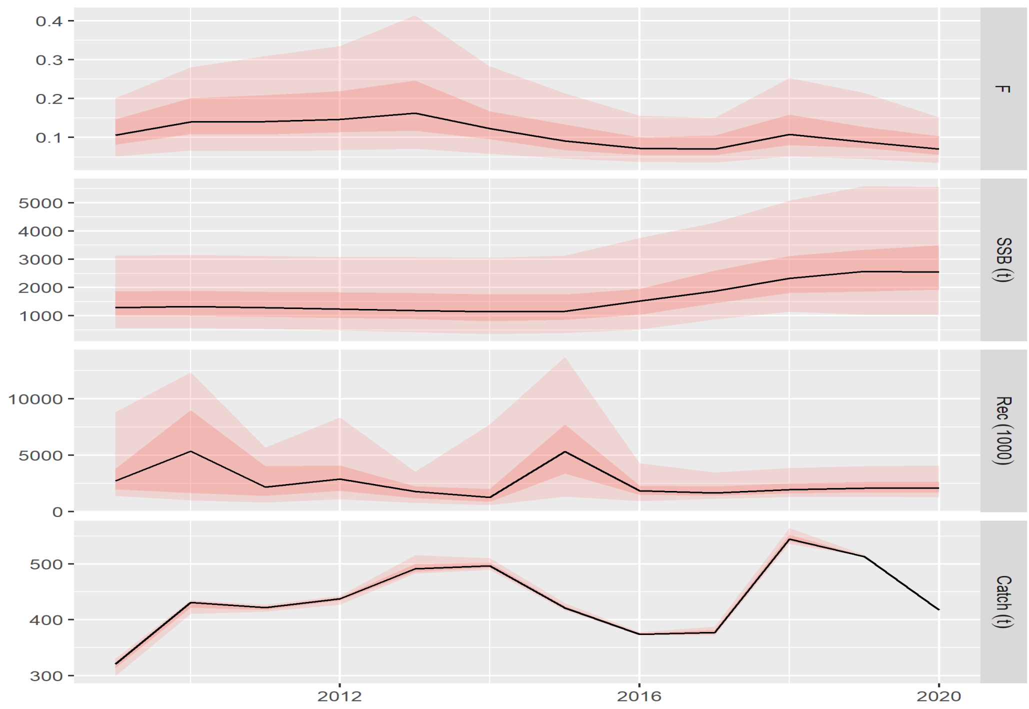

Figure F.12: Assessment summary plots of thornback ray (*Raja Clavata*) in subarea 8 using Stock Reduction Analysis (SRA) approach. Their 5%, 25%, 50%, 75% and 95% quantiles are shown as a black line (median, 50% quantile) and red ribbons.

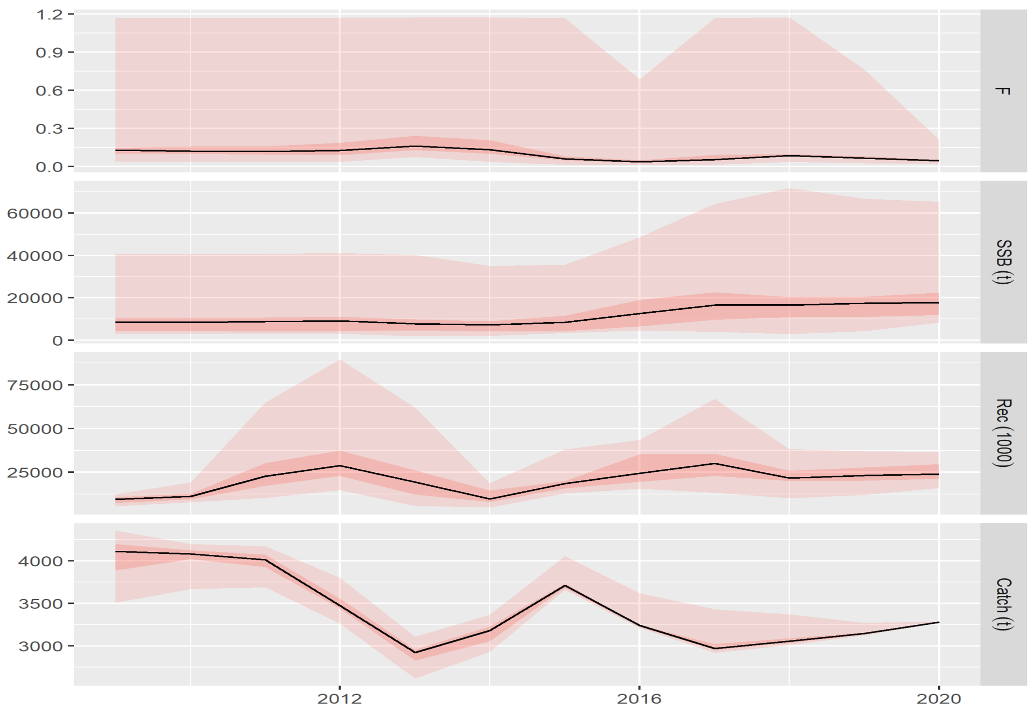

Figure F.13: Assessment summary plots of cuckoo ray (*Leucoraja naevus*) in subareas 6, and 7, and divisions 8abd using Stock Reduction Analysis (SRA) approach. Their 5%, 25%, 50%, 75% and 95% quantiles are shown as a black line (median, 50% quantile) and red ribbons.

Retrospective analysis A retrospective analysis was conducted to assess the impact of sequentially removing the most recent five years of data on the estimates of fishing mortality, spawning stock biomass, spawning stock biomass depletion and recruitment in catshark (Figure F.14), red mullet (Figure F.15), thornback ray (Figure F.16) and cuckoo ray (Figure F.17).

The evaluation of sequentially removing the most recent five years of data revealed significant disparities in the estimates of fishing mortality, spawning stock biomass, spawning stock biomass depletion, and recruitment for catshark, red mullet, thornback ray, and cuckoo ray. These disparities emphasize the sensitivity of the SRA model to the inclusion or exclusion of recent data, indicating potential impacts on the accuracy of stock assessments. Based on the results obtained in Chapter 3, the length of the data time-series used for assessing the population dynamics of these stocks would be sufficient to obtain accurate stock status and fleet level estimates from the SRA model. However, the observed variations found in the retrospective analysis highlight the necessity for further research and analysis to gain a deeper understanding of the underlying factors contributing to these disparities. Addressing these discrepancies is crucial to improve the reliability and robustness of future assessments.

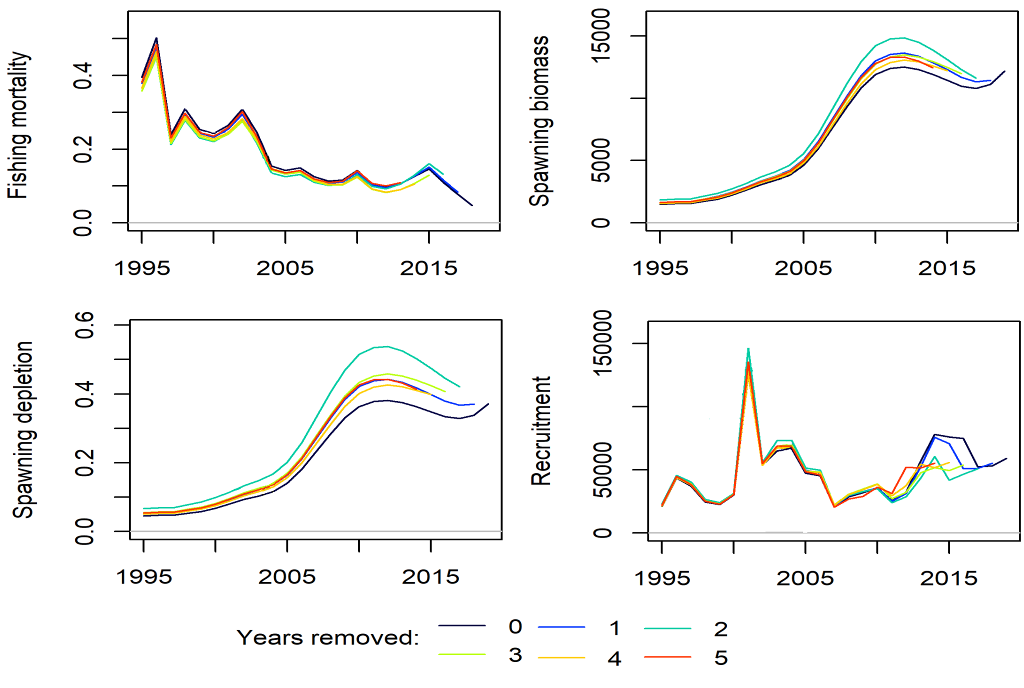

Figure F.14: Retrospective analysis of catshark (*Scyliorhinus canicula*) in divisions 8abd assessment results.

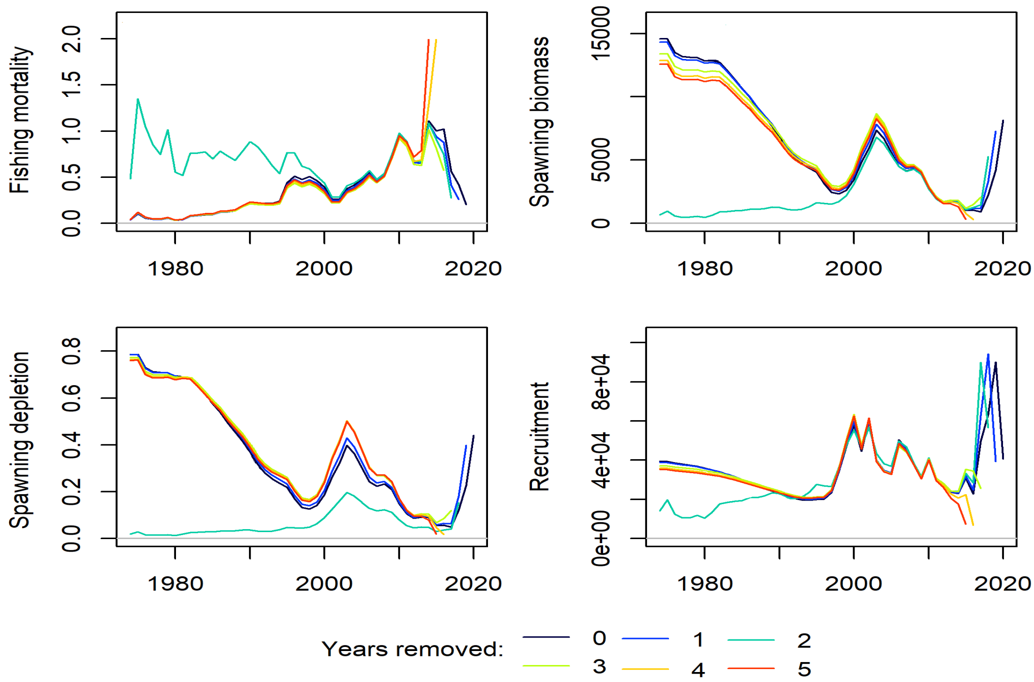

Figure F.15: Retrospective analysis of red mullet (*Mullus surmuletus*) in subareas 6 and 8, and divisions 7a–c, 7e–k, and 9a assessment results.

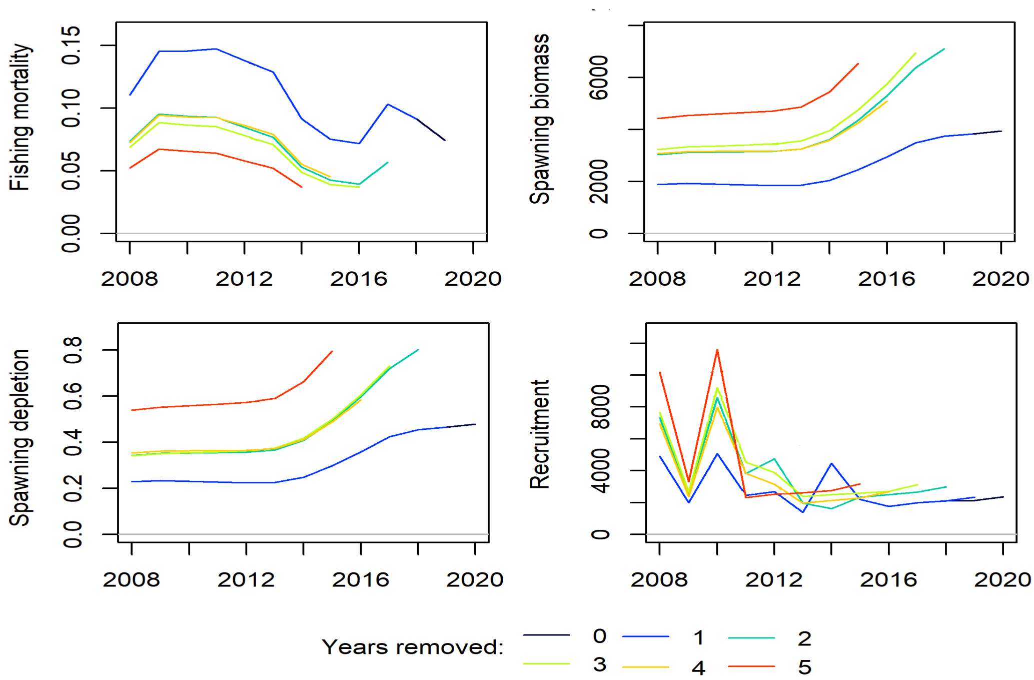

Figure F.16: Retrospective analysis of thornback ray (*Raja Clavata*) in subarea 8 assessment results.

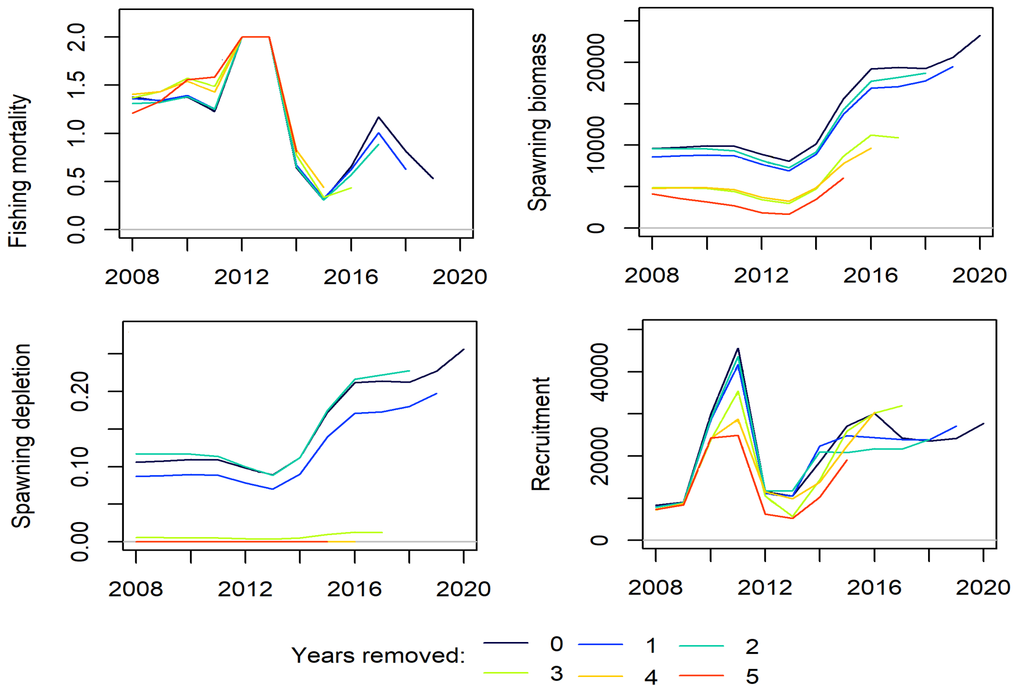

Figure F.17: Retrospective analysis of cuckoo ray (*Leucoraja naevus*) in subareas 6, and 7, and divisions 8abd assessment results.

F.1.2 Fixed population

The rest 15 out of 28 stocks data availability was not enough to apply conventional assessment model or neither the SRA model (Chapter 3). Thus, these stocks (i.e., European squid, pouting, tope shark, European conger, john dory, cuttlefish, argentine, smooth-hound, tub gurnard, whiting, poor cod, grey gurnard, undulate ray, pollack, and Norway lobster) were included in the simulation using constant CPUE approach (Table 4.2). This setting only needs catch data time-series. The biomass is assumed enough high to support the catches over time. The role of these stocks in the simulation is to restrict the fishing activity and contribute to the revenue of the fleets. We could not evaluate the effect of different management measures in the status of these stocks.

Among these stocks, Norway lobster is classified in ICES category 1, so it has an analytical assessment from ICES (ICES, 2021h). However, its assessment is based on a underwater TV survey, and it was not possible to translate the underwater TV survey to an age-structured population dynamics model. Thus, in this study, Norway lobster was included in the simulation model using a fixed population approach (i.e., CPUE approach).

Starry smooth-hound and common smooth-hound are in ICES category 3. According to Chapter 4, their data availability did not fulfill the minimum data required to obtain reliable population dynamics estimates from the application of the SRA model. Their catch and abundance index data did not cover a stock generation time, and age or length-structured data is not available for these stocks. These two stocks catch data are kept at genus level (smooth-hound, *Mustelus spp.*). Thus, in the simulation they were included at genus level and using a fixed population approach.

For stocks in ICES category 5 and 6 only catch or landing data time-series are available, they do not have any reliable index data. At the moment, there is not any method to assess their age-structured population dynamics so in the simulation model they were included using the constant CPUE approach.

F.2 Operating model: Fleet dynamics

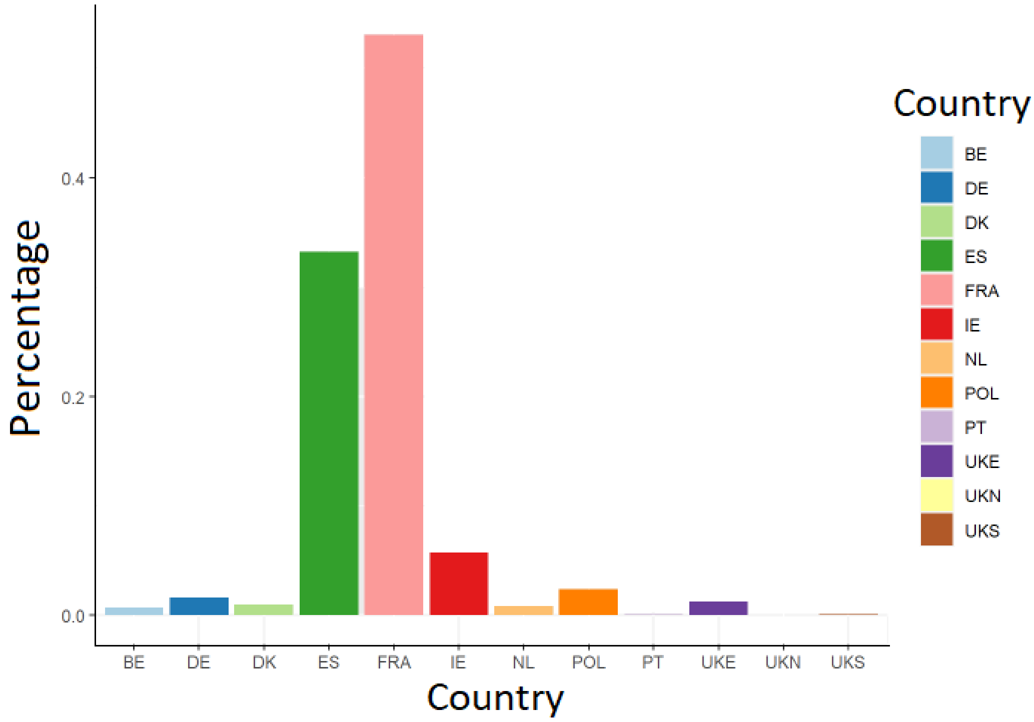

Figure F.18: Percentage of total catch in the Bay of Biscay by country. Countries accronimes; Belgium (BE), Germany (DE), Denmark (DK), Spain (ES), France (FRA), Ireland (IE), Netherlands (NL), Poland (POL), Portugal (PT), and United Kingdom (est, north and south UK, UKE, UKN and UKS respectively).

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Carlos Duarte et al. (2020), Rebuilding marie life —
We are at a point at which we can choose between a legacy of a resilient and vibrant ocean or an irreversibly disrupted ocean, for the generations to follow.

The demersal mixed-fisheries operating in the Bay of Biscay catch more than 150 species, but only eight of them are included in the Western Waters and adjacent waters multiannual management plan. This raises the issue of whether the current multiannual management plan is sufficient to ensure the sustainability of the whole system.

In this PhD Thesis, we evaluated the impact of current and alternative management strategies on the sustainability of the system in a simulated reality. In comparison to previous works, we broadened the scope of the work to non-target and data-limited stocks. First, we applied a stock prioritization approach to identify the stocks most at risk, most exploited, and/or economically most important for the fishery. The potential risk was calculated using the Productivity-Susceptibility Analysis (PSA). Then, we evaluated by "self-test" simulation the ability of a low data-demand assessment model like Stock Reduction Analysis (SRA) to provide accurate population estimates under alternative data availability, population exploitation levels and initial population assumptions. This allowed us to identify the data-limited stocks whose dynamics could be reliably mimicked based on the SRA model. Finally, we evaluated the performance of current and alternative management strategies. The simulation included 28 stocks: 13 had age-structured population dynamics (9 data-rich stocks based on analytical assessments and 4 data-limited stocks using SRA model) and for 15 stocks the catch was proportional to the effort level and independent of their abundance.

The obtained results contributed to providing a holistic evaluation of the management of the mixed-fisheries operating in the Bay of Biscay.