



**Department of
Mathematics and
Statistics**

MARINE MODELLING RESEARCH REPORT

March 2020

**Risks to North Sea fish
stocks and wildlife if post-
Brexit fishery negotiations
fail to reach agreement on
quotas and access to UK
waters**

**EXTENDED TECHNICAL
REPORT**

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Published in March 2020 by The University of Strathclyde, Glasgow, UK

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Available for downloading from:
<https://doi.org/10.17868/71708>

A shorter, summary version of this report is available from:
<https://doi.org/10.17868/71709>

Suggested format for citation:
Heath, M.R. & Cook, R.M. 2020. Risks to North Sea fish stocks and wildlife if post-Brexit fishery negotiations fail to reach agreement on quotas and access to UK waters: EXTENDED TECHNICAL REPORT. University of Strathclyde, March 2020. 136pp.
<https://doi.org/10.17868/71708>

Risks to North Sea fish stocks and wildlife if post-Brexit fishery negotiations fail to reach agreement on quotas and access to UK waters: EXTENDED TECHNICAL REPORT

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SUMMARY AND KEY FINDINGS

“Taking back control of fisheries” became one of the totemic issues uniting supporters of the campaign to leave the EU. Having left, the issue is again high on the agenda in the ‘future relationship’ negotiations. The UK Government has indicated that getting a better deal for UK fishermen is a “red line” in the negotiations. This includes increases in quota for UK vessels, and restrictions on access to UK waters by foreign vessels. However, the EU has linked access to UK waters and maintenance of quotas enshrined in the Common Fisheries Policy (CFP) with securing tariff-free trade in fish and other products.

This report focuses on the North Sea and provides an assessment of the risks to stock and ecosystem conservation associated with the post-Brexit fisheries negotiations. The report first sets out the history behind the allocation of quota shares (Relative Stability) and compares the UK shares with those under proposed alternative rules based on the distribution of fish (“zonal attachment”). Unless a negotiated agreement can be reached to resolve these different views on quota allocation there is a risk that unilateral actions will result in the combined catches by all states exceeding the levels required for long-term maximum sustainable yields. The report sets out a narrative for the impact of such unilateralism on harvesting rates, and then presents results from models which show the risks that these would pose for key fish stocks and wildlife.

Key findings of the report

- It is the case that other EU and neighbouring nation states take more fish overall from UK waters than UK vessels take from theirs. However, this discrepancy is not universal across all fish stocks or regions.
- A combination of Relative Stability and national and international quota swaps and trades leave the UK with a preferential proportion of the quota for some species but an unfavourable proportion of others. For example, the UK took, on average, 87% of

the haddock landings from the North Sea during 2003-2013 which exceeds current estimate of the zonal attachment based quota.

- For other stocks such as scallop, crabs and lobster which are valuable to the UK, the CFP and Relative Stability plays little or no role as the fisheries are carried out almost exclusively within existing national jurisdictions.
- In fact, a high proportion of the apparent disparity between total UK share of catches, and reliance of EU vessels on UK waters, is due to just four species – herring, mackerel, blue whiting and sandeel.
- In the event of post-Brexit unilateralism on quota setting the likely outcome is increased overall fishing levels that would more than double the risk of overfishing herring, and increase the risk of North Sea cod stock collapse by 75% (with stock levels falling below sustainable limits).
- Reduced food abundance and increased by-catch in fishing gear in a post-Brexit unilateralism scenario are likely to be the main risks for wildlife, resulting in a projected decline in cetaceans and seabirds.

Key conclusions from the report

Overall, the report demonstrates the importance of international adherence to scientific advice on catch limits for shared fish stocks. It is vital that the UK, EU and other coastal states such as Norway reach an agreement that that preserves cooperation on fisheries, even in the face of political pressure to act independently for short term gain. Otherwise, the outcome will be overfishing, eventual collapse of key stocks, declines in seabird and cetacean numbers, placing further pressure on the UK's coastal communities that rely on a healthy marine environment to survive.

The potential risk to North Sea cod stocks identified in the report is of particular concern. The modelling indicates a 75% risk of cod stocks falling below sustainable limits if the UK and other nation states set their quotas unilaterally.

It is also concerning that a unilateralist outcome is expected to lead to declines in seabird and cetacean numbers. The UK Government has identified the conservation of marine wildlife for future generations as a key action. These populations are already under pressure from climate change and a diversity of human activities so it would be detrimental to add further pressure from overfishing.

The findings highlight the need for the UK to have an effective, independent environmental watchdog to ensure fishing limits are set at sustainable levels.

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1. Background and context

The UK left the European Union on 31 January 2020 under the terms of the “New Withdrawal Agreement” [1]. This marked the start of a transition period during which the future relationship between the UK and EU will be negotiated, as outlined in the associated “Political Declaration”.

The Political Declaration makes specific provisions for the departure of the UK from the European Common Fisheries Policy (CFP) which regulates fisheries activities across European waters [2, 3, 4, 5, 6, 7, 8, 9]. The Declaration states that new arrangements are to be sought regarding access to waters and quota shares, while at the same time ensuring fishing at sustainable levels to “promote resource conservation, and foster a clean, healthy and productive marine environment”. The Declaration envisages that these negotiations will be concluded by 1 July 2020 (Political Declaration page 14, Section XII; [1]).

Freedom of access to the UK Exclusive Economic Zone (EEZ) for foreign fishing vessels and objections to the CFP ‘Relative Stability’ rules were among the high-profile arguments for exiting the EU [3, 10, 11, 12]. Relative Stability is the fixed allocation key which, since 1983, has governed how the Total Allowable Catch (TAC) for each of the EU-managed fish stocks is shared out between the fishing industries of the Member States. The amount allocated to each Member State is referred to as ‘quota’ [13, 14, 15, 16, 17]. The UK fishing industry has argued that Relative Stability has become grievously unfair to the UK and should be re-negotiated, advocating that fishing opportunities for shared stocks should be divided according to ‘zonal attachment’ – of which there are several definitions but is generally taken to refer to the proportional distribution of stock between EEZs [8, 18, 19, 20, 21]. On this basis, it is argued that the evidence suggests that the UK should receive a larger quota, or proportion of the available TAC, for many of the species and stocks in its own EEZ.

It is clear that access to UK waters for EU vessels and the allocation of quotas will be a key strand of the transition-period negotiations regarding future relations [22, 23, 24, 25]. For example, the briefing document for the Lords stages of the UK Fisheries Bill 2020 states that:

"The EU is seeking continued reciprocal access to fishing waters and stable quota shares for the amount of fish that can be caught. The UK argues British fishing waters should primarily be for British fishing vessels, and opportunities for EU vessels should be negotiated annually on the basis of scientific data about sustainable catch levels, not based on historic quotas." [26].

There is thus a potential scenario under which there is no agreement on these issues before the deadline set by the Political Declaration, or at least not to the satisfaction of the UK fishing industry [8, 24, 27, 28, 29, 30, 31, 32, 33, 34]. The plausible worst case consequence would be that the UK is unable to effectively police harvesting in its EEZ after the transition period [35] and/or ends up unilaterally setting its own quotas within its EEZ in excess of Relative Stability. Given domestic pressure the UK’s aim (officially sanctioned or otherwise)

would presumably be to take what it regards as its perceived “fair share” of the TAC for shared stocks [4, 26], although this carries the risk of retaliatory tariffs on exports of fish products to the EU [15, 23, 36, 37, 38]. If the remaining EU Member States adhere to their existing Relative Stability quota allocation then, instead of being harvested within the TAC limits advised by the International Council for the Exploration of the Sea (ICES), these stocks will be over-exploited.

It is to be expected that unilateralism as outlined above would have detrimental consequences for fish stock conservation and the principle of harvesting at Maximum Sustainable Yield (MSY – the largest yield (or catch) that can be taken from a species’ stock over an indefinite period). There would also be consequences for wildlife that depend on there being sufficient fish in the sea to feed. However, there has been no risk assessment of these impacts of post-Brexit fisheries negotiations breaking down into unilateralism. The purpose of this report is to provide an assessment of these risks for the North Sea.

1.1. Structure of the report

The report:

1. Briefly explores the history of internationally agreed TACs in the North Sea, including the factors behind the setting of TACs prior to and after implementation of the Common Fisheries Policy (CFP). Background is provided on why the North Sea is so important to the UK and EU fishing industries and potential impacts of overfishing the stocks.
2. Briefly summarises the zonal attachment analyses that have been published to-date, in the context of Brexit. New analysis of the distribution of landings based on the FAO/ICES and EU Scientific, Technical and Economic Committee for Fisheries (STECF) landings databases are used to establish potential scenarios for post-Brexit negotiations on quota share for North Sea shared stocks.
3. Looks at the consequences for key commercial stocks under different scenarios – i) where stocks are managed in line with current TAC advice in accordance with MSY principle. This scenario represents both continuation of Relative Stability or an amicable outcome of quota reallocation negotiations so that existing TAC advice continues to be respected; ii) a post-Brexit scenario where the UK fishing industry does not adhere to Relative Stability quotas and instead take what they consider to be their fair share of each stock as identified under zonal-attachment considerations, and where neighbouring Member States are unwilling to give up their current Relative Stability quota, leading to TAC overshoot.
4. Models of the consequences for the wider ecosystem of the negotiation-breakdown scenarios with an explanation of the potential consequences of these impacts including on cetaceans and seabirds.

1.2. Review of the issues

Sea fisheries issues formed a significant part of the persuasive argument in favour of the UK deciding to leave the EU in the lead-up to the Referendum in 2016 [10, 11, 12, 30, 39, 40, 41, 42]. The infamous statistic, repeated many times during the Referendum campaign – that 80% of UK fish is given away to the rest of Europe – is mirrored in the Forward to the White Paper on Defra’s vision for a future fisheries policy once the UK is no longer part of the EU :

“On average between 2012 and 2016 other EU Member States’ vessels landed in the region of 760,000 tonnes of fish (£540 million revenue) annually caught in UK waters; whereas UK vessels landed approximately 90,000 tonnes of fish (£110 million revenue) caught in other Member States’ waters per year in the same time period” [3].

Similarly, the briefing notes for The Queen’s Speech, December 2019, state that:

“On average annually between 2012 and 2016, other EU Member States’ vessels landed in the region of 749,000 tonnes of fish (£575 million revenue) caught in UK waters. UK vessels landed approximately 96,000 tonnes (£96 million revenue) caught in other Member States waters per year in the same period” [43].

The White Paper [3] cites this as evidence that “... the UK’s share does not accurately reflect the resources in UK waters”, and that the “... allocation of fishing opportunities under the CFP is outdated”. Further it is stated that “As we leave the EU, we are committed to working closely with our partners to manage shared stocks in a sustainable way and share fishing opportunities on a fair and scientific basis.”

Taken together, the landings-based evidence cited above and government statements of intent, have raised expectations in the UK fishing industry and the national perception, that UK fishing communities can expect a significant rise in revenue and employment as a result of the UK leaving the EU [31, 32, 33, 44, 45, 46, 47]. These perceptions have been reinforced by economic analyses based on the assumption that the UK will be able to take back substantial proportions of the annual TAC (quota) for key stocks around the UK from its erstwhile partners in the EU (e.g. [38, 48]. Walmsley *et al.* (2018) [38] foresees a £540 million increase in economic output and 5,000 extra jobs for seafood industries, given a favourable resolution of quota and tariff negotiations

However, other commentators urge caution over the anticipation of economic benefits, pointing out that concentration of a majority UK quota for some species is in the hands of a small number of powerful vessel owners means that any benefits from Brexit will be very unevenly distributed [23, 49]. In addition, subsequent analysis of media coverage of fisheries issues during the Referendum Campaign indicates bias towards offshore fishing interests that stand to benefit from any increases in quota, and that the views of smaller scale inshore fisheries were under-represented [50]. For the inshore fleets, Brexit offers little or no quota

opportunities while tariff issues pose a potential threat. Similarly, the risks that Brexit could precipitate overfishing were hardly covered [51].

In addition, others note that the UK will face some difficult challenges in reaching an agreement with the EU on quotas and access to each other's waters after Brexit [52]. First and foremost, the EU can obviously be expected to resist strongly any attempt to change Relative Stability to its detriment [24]. It is argued that the UK holds many 'trump-cards' in terms of access to UK waters for other nations to be able to catch even a reduced quota. However, while the Scottish fishing fleet depends relatively little on non-UK waters, the English fleet exploits Irish, French and Norwegian, as well as UK, waters. In particular, UK vessels benefit from rights to fish for cod in North Norwegian waters in return for transfers to Norway of quota for stocks which are mainly of value to other EU countries. Maintaining this arrangement will be a difficult negotiation. Finally, the UK fishing industry depends heavily on exports to the EU, so is vulnerable to tariffs. The value of the UK's fish exports to the rest of the EU (including farmed salmon) exceeds the quayside value of landings, and the 2018 Marine Scotland report ([38] Walmsley *et al.* 2018) highlights that any anticipated benefits arising from quota negotiations are contingent on tariff free or low-tariff trading [7]. Countries which stand to suffer most from a renegotiation of Relative Stability in the UK's favour, such as France and Spain would have every incentive to demand high tariffs on fish imports from the UK.

Some sections of the UK industry advocate a 'nuclear option' if quota negotiations go badly, in which the UK stakes its own claim to a share of the TAC for each stock without the agreement of remaining EU nations and Norway [8, 41, 51]. However, there would clearly be consequences in terms of tariffs, the ability of the UK to police its waters against vessels from the rest of the EU, and in addition the likelihood of overfishing and the depletion of stocks. It is this latter aspect that we seek to explore in this report.

1.3. The case for re-negotiating Relative Stability

The simple infamous statistic, repeated many times during the Referendum campaign and again (essentially) in the Forward to the White Paper [3] and the December 2019 Queen's Speech [43] – that 80% of UK fish is given away to the rest of Europe – is dramatic and conveys the impression that UK is unfairly treated by the quota system at the heart of the CFP. The argument is that Relative Stability – the fixed allocation key (quota system) by which the agreed Total Allowable Catch (TAC) for each stock is dis-aggregated and allocated to member states – is no longer appropriate for a variety of reasons.

Relative stability was negotiated in the early stages of the CFP, which was ratified in 1983. Based on the principle of shared access to each other's waters, the CFP established a formula for sharing out an annually agreed TAC in proportion to national landing statistics during the period 1973-1978. Since then, the relative abundances of fish species have changed and fish distributions have shifted northwards partly due to warming sea temperatures [53], so that

UK waters now harbour a greater proportion of the available fish than in the past. However, UK vessels, which mainly operate in UK waters, are not able to exploit this opportunity due to the fixed quota system.

Implementation of the EU Landing Obligation, aiming to eliminate the wasteful practice of discarding at sea, has compounded the issue of Relative Stability. Discarding restrictions have led to the emergence of so-called ‘choke species’ for which the UK industry has only a small quota, but now constitute an increasing proportion of catches in mixed species fisheries. The prime example is hake in northern UK waters, where vessels find it difficult to take their allocation of other demersal species such as haddock and whiting, without exceeding their hake quota and risking illegal discarding [54]. Work is on-going to reduce the impact of choke species, through improved gear and spatial selectivity. Even though full implementation of the Landing Obligation occurred in January 2019, there is anecdotal evidence of on-going non-compliance [55].

There are many possible objections to the basis on which Relative Stability was established, in particular the notion that historic landings are a valid basis for asserting ownership of harvesting right. However, before considering these, it is important to appreciate the context in which the CFP was established. During 1939-1945, when Europe was at war, any sort of large scale fishing was almost impossible due to risk of attack and requisition of vessel. So, this was a period of very low fishing mortality and the evidence suggests that stocks expanded significantly as a result [56]. Immediately after the end of the war, the UK developed distant water fleets to harvest cod in the Arctic and around Iceland, while Norway and other nations rapidly expanded fleets to take advantage of the stocks in European waters. Harvest from the North Sea increased dramatically, partly on the rebound from the low fishing mortality during the war-years, and partly due to a series of exceptional recruitment events of cod and haddock possibly associated with a sequence of cold-winter years in the 1960’s. Norway, and for a brief period the USSR took extraordinary landings of cod and haddock from the North Sea during this period. However, in the absence of any effective coordination of fisheries management or adherence to agreements on harvesting limits, the rise in exploitation was essentially mining out the stocks. By the time the UK fleet had lost its conflict with Iceland over access to waters in their EEZ in the 1970’s, fisheries in the North Sea were starting to decline and stocks were showing clear signs of over-exploitation. The UK had essentially ‘missed the boat’ in regard to establishing a presence in the North Sea. So, the CFP was established in the context of stocks in crisis, and without any coherent internationally-based management. Against that backdrop, negotiating Relative Stability was a significant achievement [57, 58, 59].

Since 1983, the CFP has successfully reined-in the over-capacity of the 1970’s and restored order to fisheries in its northern waters at least. Many stocks are now operating at around their MSY and classified as sustainable, though the public perception is still that fish stocks are in decline and that the CFP is to blame [60, 61, 62]. Hence, the context today for a renegotiation of Relative Stability is completely different from that which prevailed when the rules were originally established. We should probably see Relative Stability as having been a

critical factor in establishing order in the 1980's, but has now run its course. The question is what to replace it with, and arguments on this seem likely be just as intense as they were in 1983.

The original argument for establishing Relative Stability on the basis of landings history was presumably that there were no other viable data available at the time for establishing the extent of each nation's rights to the stocks. However, the distribution of landings reflects not just the distribution of fish but also the capability and economic legacy of each nation during the census period, the economics of the fishing operations. Today, there is the additional complication that landings are strongly limited by quotas, so in no sense do they measure the full capacity of a nation to exploit stocks in its usual fishing grounds, though it is argued that quota-trading and swaps act to reduce the distortion of catching capacity caused by restrictive quotas [63]. Obviously, these restrictions were much less of an issue in 1983 at the start of the CFP. However, the question that Relative Stability was attempting to solve in 1983 was rather different from the issue today. Then, the objective was a fair sharing out of a limited and dwindling resource in the region as a whole, assuming free and open access to all members' waters and taking into account the economic needs of coastal areas, including those affected by the loss of fishing opportunities arising from the conflict with Iceland, and national priorities in terms of target stocks [64]. Today, faced with the departure of the UK from the EU, the question revolves around the proportion of each shared stock that the UK is entitled to claim as its own based on occupancy of its EEZ, or the contribution that its EEZ makes to the wellbeing of the stock – equivalent arguments to those employed by Iceland in relation to its unilaterally established 200 mile fishing exclusion zone up to 1976.

As an independent coastal state, the UK will be legally entitled to control access to its waters by all foreign vessels seeking to harvest fish stocks, even those which are deemed to be a shared resource since their distribution and migrations extend beyond that of the UK EEZ [28, 65, 66]. On the other hand, the UK is required under the United Nations Convention on the Law of the Sea (UNCLOS) to cooperate with other relevant nations in the management of such shared stock and the setting of a TAC, and should permit access for other nations to take any harvest allocation (quota) that it is unable to utilise itself [67]. However, there seem to be no hard and fast rules for establishing a quota allocation in these circumstances, so the issue is determined by a negotiation [68].

Assuming that the existing EU-Norway negotiations on access rights and quota allocation of shared stocks are a credible model for how this process might operate, the steps involved are that the parties agree on a TAC, then on a zonal attachment of the stock (i.e. what proportion is associated with each EEZ), and finally on a quota share of the TAC (e.g. [69]). The quota assignment is not necessarily the same as the zonal attachment and can take account of any factors that the parties agree to consider, including legacy fishing patterns, economic considerations, investment in research and monitoring to support the stock assessments, trading of quota for access to other resources. It is clear from the arguments above that recent catch (landings) distributions are probably a poor basis for establishing zonal attachment, but

certainly have a role on the negotiation of how quota might vary from zonal attachment, and on access rights.

The most straightforward approach to zonal attachment is to consider the spatial distribution of stock biomass. The technicalities of deriving this from scientific surveys are not trivial, but the essence is to determine the proportion of time that the stock biomass spends in each EEZ over its life cycle. In the North Sea, a protocol and equations for this process, based on the annual ICES coordinated International Bottom Trawl Survey (IBTS) data, was set out by Needle (2015) [17], and extended in a report for the SFF by the University of Aberdeen (2017) [18, 21]. Variations on this basic concept include assessment of the spatial distribution of biomass production, i.e. the geographic areas where the biomass accumulates as a result of food consumption and growth – this is the production which is available to support a sustainable harvest without depleting the stock [9]. Production and harvest need not occur in the same locations especially for highly migratory species. Nevertheless, analysis of the spatial distribution of landings continues to be confused with the concept of zonal attachment [48], 3], and note that the infamous statistic upon which the fisheries case for leaving the EU was made, was based entirely on landings data.

In the remainder of this report, we identify some likely quota-goals that the UK might adopt in negotiations with the EU and Norway regarding North Sea stocks of various fish species, based on analyses of survey biomass distribution to establish zonal attachment, and landings data to set the context in terms of historic activity. We then use these to scope some plausible narratives for a breakdown of negotiations between the UK, EU and Norway on fisheries in the North Sea and a descent unto unilateralism. Based on these narratives, we then assess the consequences for stock conservation and the status of the ecosystem, including wildlife which depends on fish and fisheries.

1.4. Scope of the study

Our study has four main parts:

1. An analysis of different sources of historic landings, relative stability quota shares, and estimates of zonal attachment based on survey data. The landings data came from two sources: the official statistics published by the FAO and ICES [70], which resolve landings of over 500 different species by individual nations since 1903, but only by coarse ICES fishing areas; and the STECF database of effort and landings [71], supplemented by data from the Norwegian Fisheries Directorate, which contains data by individual statistical rectangles (1 longitude x ½ latitude – approximately 30 x 30 nautical miles), but only since 2003. Details of data and methods are contained in Appendix 1. Data on Relative Stability quota shares of all the species managed by the EU on the shelf waters around the UK were extracted from the Official Journal of the European Union, publishing the Regulations setting out the annual fishing opportunities in Union waters. Estimates of zonal attachment were compiled from two recent publications: The Annexes to the White Paper on Defra’s vision for a future

fisheries policy once the UK is no longer part of the EU [3], and a report by Aberdeen University for the Scottish Fishermen's Federation [21]. The purpose here was to set in context the benefits accrued from different fish species in the waters around the UK, and how the UK's share of these benefits has changed over time.

2. Based on our analysis of landings data, Relative Stability quota shares, and the published works on zonal attachment of fish species, we developed narratives for unilateralism scenarios of the outcome of UK-EU-Norway shared-stock quota negotiations. In these, the UK claims, but fails to secure its quota goals for North Sea fish stocks because the EU and Norway refuse to yield quota and accommodate the UK's demands. As a result, the UK unilaterally harvests its claimed share of the TAC, while the EU27 and Norway continue with their status-quo Relative Stability quota. As a consequence, overall fishing mortality rates increase from present-day values. Note that we are not concerned here with the benefits or detriments to either the UK or the rest of the EU which might arise from either a successful or an unsuccessful negotiation. Our aim is to assess the consequences of an unsuccessful negotiation for fish stocks and the ecosystem. We can take it as read that a successful outcome maintains the status-quo TAC regime aimed at achieving MSY, and merely redistributes quota with no consequences for either stock conservation or the ecosystem.
3. For our unilateralism scenario we ran a) stock projection models for the main individual species in the North Sea based on an implementation of the medium-term forecast models employed by ICES, and b) projections of the outcome for the North Sea food web as a whole, including the collateral effects on the seabed and non-target wildlife, using the StrathE2E marine ecosystem model [72, 73, 74].

2. Long-term trends in landings and the UK's share of landings for waters around the UK

The infamous fisheries statistic used to make the case for leaving the EU – that 80% of UK fish is given away to the rest of Europe, or that other EU Member States' take 8-times more fish from UK waters as we take from theirs – is based on an analysis of the combined landings of all fish species, in all the European waters where UK vessels operate. To put this statistic into context, we aggregated the total international landings of all the species in the ICES/FAO landings data base over the five fishing areas around the UK (FAO areas 27.4 (North Sea), 27.6a (West of Scotland), 27.7a (Irish Sea), 27.7d,e (English Channel), and 27.7b,c,f,g,h,j (Celtic Sea); Fig. 1), and apportioned these between the UK, the EU, and other nations, over the period 1903-2016. The results (Fig. 2) show that while 50-60% of all landings from these five regions combined currently (since 2003) come from UK waters, the UK has accounted for only 20% of the total international landings since joining the EU. However, this has not always been the case. At the beginning of the 20th century the UK landed around 80% of all fish and shellfish, but this proportion declined steadily over time, especially between the two World Wars. In fact, after joining the EU, the UK's share of

landings recovered slightly and has stabilised since coming under the CFP Relative Stability rules. The Relative Stability proportion of total landings allocated to the UK is difficult to establish for the aggregation of species presented here; since the quotas are set on a species-by-species basis the group quota proportion will vary from year-to-year as the TACs for individual species vary. Hence, in Figure 2 we show the Relative Stability fraction only for 2018. In addition, there are many species included in the total landings records that are not subject to CFP TACs or quotas, though these make up only a small fraction of the total, at least for finfish. For example, many shellfish TACs are set and managed under national jurisdictions and do not form part of the CFP infrastructure since the fisheries are carried out within 3 or 12 nautical mile limits.

Analysing the STECF data for 2003-2013 to determine which nations are fishing in each EEZ zone (Fig. 3), reveals that the UK accounted for around a third of all the landings from the UK EEZ, and re-affirms the infamous statistic – the data from the more restricted region analysed here shows that vessels from other EU states took 6.2-times as much fish from the UK EEZ as the UK takes from the non-UK parts of the region (which include Norwegian and Faroese territory). This figure increases to 8.8-times if the landings by Norwegian and Faroese vessels are included. However, breaking down the ‘all species combined’ analysis into the constituent groups of fish and shellfish types exposes important structural differences that affect the different sectors of the fishing fleets (Fig 3 and Figs A2.2-A2.5). The disparity between UK share of the total landings, and the proportion taken from UK waters is greatest for the planktivorous fish group (herring, sprat, sandeels), which make up about half by weight of all landings. In contrast, the disparity is substantially smaller for the migratory species (blue whiting, mackerel), smaller still for the demersal fish (cod, haddock, whiting, plaice etc), and negligible for shellfish (to the extent that this can be estimated). In fact, the ‘all species combined’ disparity between UK landings share and proportion taken from UK waters is due largely to the distribution of planktivorous fish landings (herring, sandeel, sprat).

Our conclusion from this section of the study is that, in so far as there is a case to be made that the Relative Stability harvest limits assigned to UK fleets are disproportionately small compared to the proportion of total harvest taken from UK waters, then this is not evenly distributed across sectors. The pelagic sector, harvesting planktivorous and migratory species, would appear to have the strongest case for a re-negotiation of quota shares. Across the range of species as a whole, the scope for gains in the demersal and shellfish sectors would appear to be substantially smaller or non-existent. However, this is not to rule out potential gains at the scale of individual species, and in individual stock areas. In the following section we give our attention to scoping the scale of adjustments to quota shares that the UK might legitimately seek as a negotiating stance, focussing on the North Sea stocks.

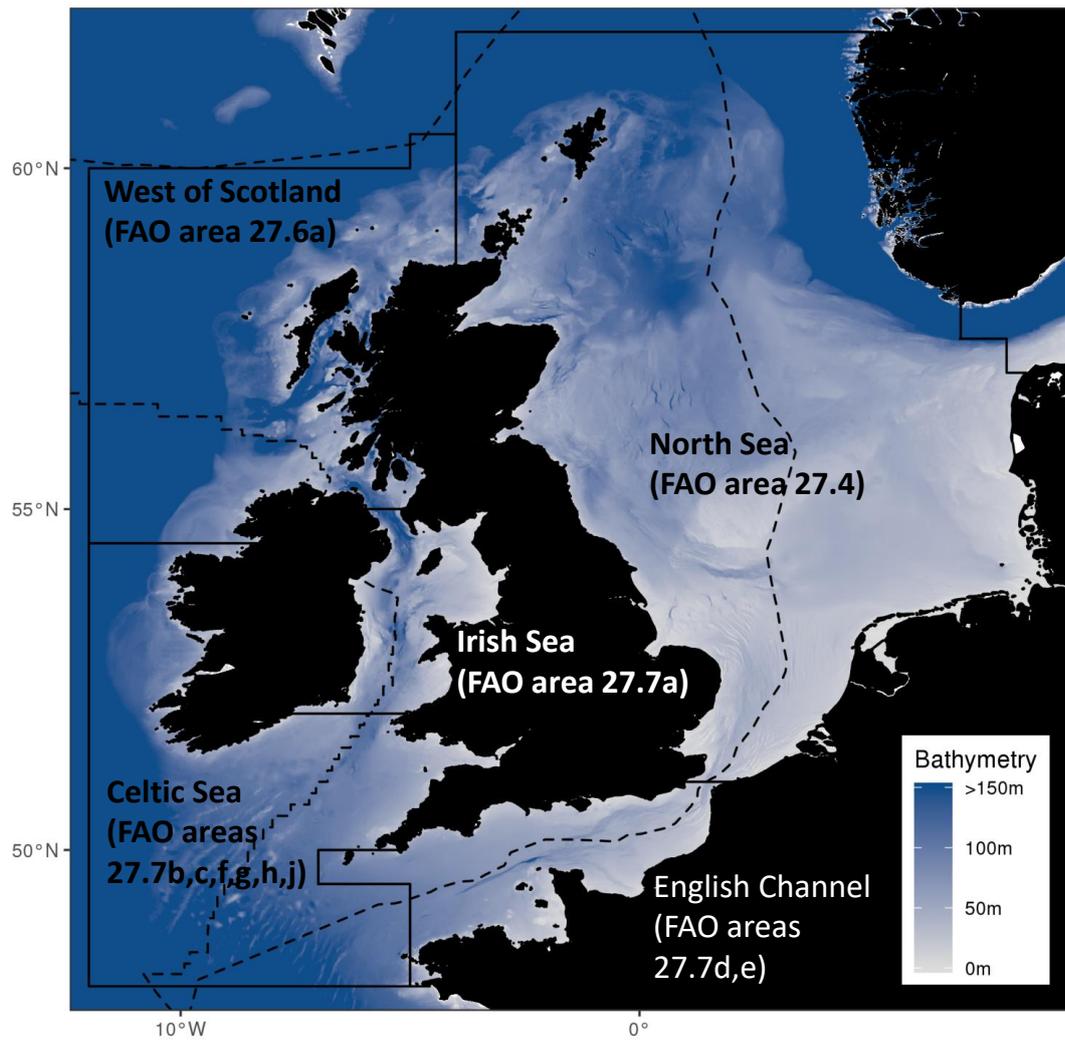


Figure 1. Map of FAO fishing areas and the UK EEZ. Solid lines show the FAO area boundaries; dashed lines show the boundary of the UK EEZ.

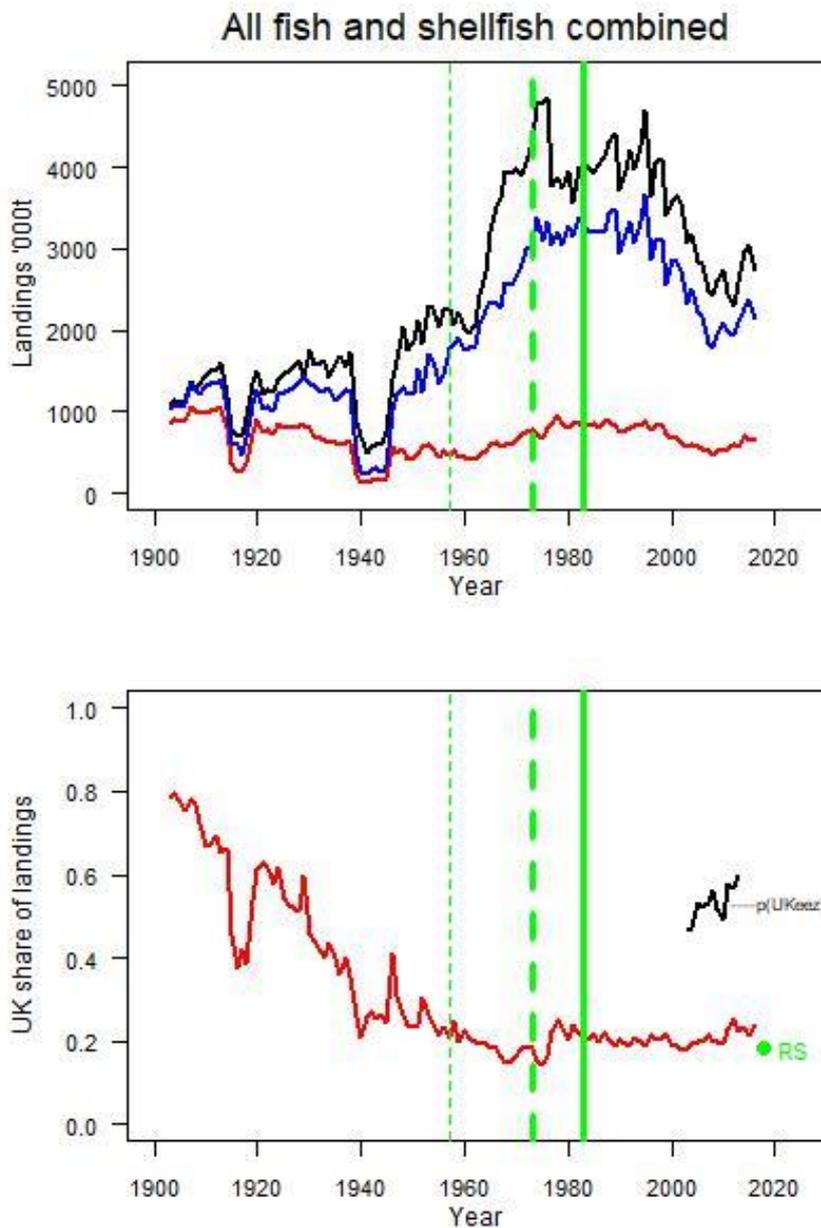


Figure 2. Landings history for all regions around the UK and all species combined. Top panel: Black line shows the total international annual landings; blue line shows the landings by the combination of all nations currently members of the EU28 (including the UK); red line shows the landings by the UK. The solid vertical green line indicates the start of the CFP; heavy vertical dashed green line shows when the UK joined the EEC, and the light vertical green dashed line shows the formation of the EEC by the Treaty of Rome. Lower panel shows the proportion of total international landings due to the UK (red line). The green symbol labelled RS indicates the UK's Relative Stability share of all EU managed species in 2018. The short black line spanning 2003–2013, labelled 'p(UKeez)' indicates the proportion of total international landings taken in the UK exclusive economic zone.

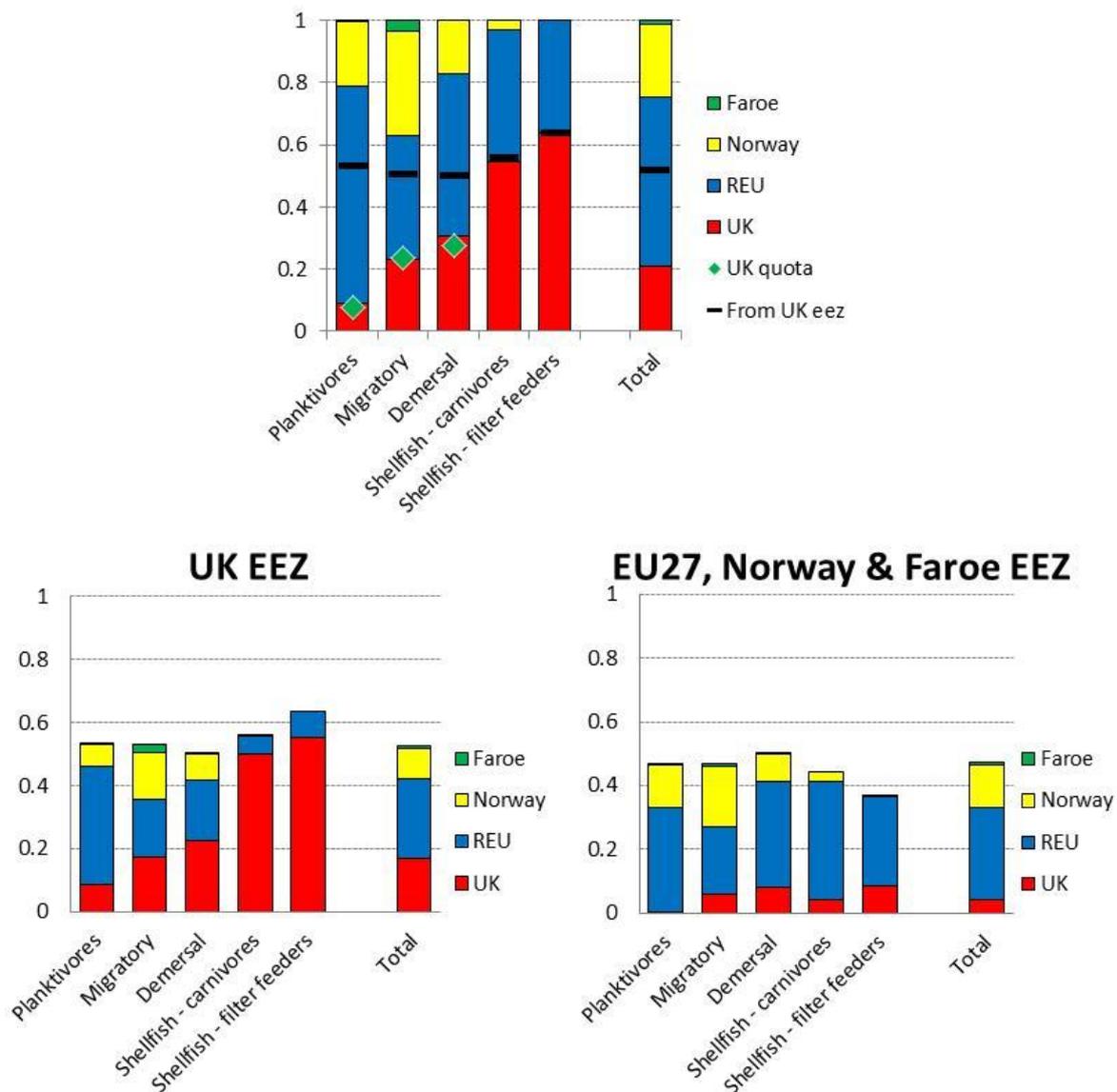


Figure 3. Average distribution of landings between nation-groups and EEZ's for all regions around the UK during 2003-2013. Top panel: Proportional distributions of landings from the region around the UK between the UK, rest of the EU (REU), Norway and the Faroe Islands. The UK quota value corresponds to 2018. The horizontal black bar indicates the proportion of landings taken in the UK EEZ. Lower panels: proportional distribution of landings in the region as a whole between nation-groups within the UK EEZ (left), and the remaining non-UK EEZs (right).

2.1. Distribution of landings and the UK's share of landings for the North Sea

Focussing on the North Sea, we repeated the analysis of landings history based on the coarse groupings of species (Fig. 4, and Fig A2.6-A2.10). These results show a very similar pattern to those for the combined sea-areas around the UK combined. This is not surprising since the

North Sea yield constitutes over half of the total landings (1.45 million tonnes per year from the North Sea compared to 2.31 million tonnes from the wider region during 2003-2013).

Around half of all fish and shellfish combined that are landed from the North Sea are taken in the UK EEZ (Fig. 4), but the UK only catches about a quarter of this. The other EU nations catch around half, and Norway the remaining quarter. The Faroe Islands take only a negligible fraction of the overall landings from the region. As a result, the so-called ‘infamous statistic’ for the North Sea is that the other EU nations take 12.9-times as much fish and shellfish from UK waters as the UK takes from theirs, increasing to 17.6-times when Norway are included. However, this disparity is even more strongly dependent on the planktivorous fish in the North Sea than in the wider waters around the UK - the infamous statistic for the planktivorous fish group is over 300-times; the UK landed on average less than 1000 tonnes of herring, sandeel, sprat and other planktivorous fish from non-UK EEZs during 2003-2013, while the other EU nations landed over 300,000 tonnes from the UK EEZ.

We examined the data for the major individual species in the North Sea from across the major groups. Working at the individual species level, the Relative Stability proportion assigned to the UK is an all-time constant fraction (since 1983), rather than a variable dependent on species compositions. In addition, for some species there are estimates of biomass zonal attachment to the UK EEZ, from the 2018 White Paper [3] and the Aberdeen University report for the Scottish Fishermen’s Federation [21]. We compare these zonal attachments with the proportional distributions of landings.

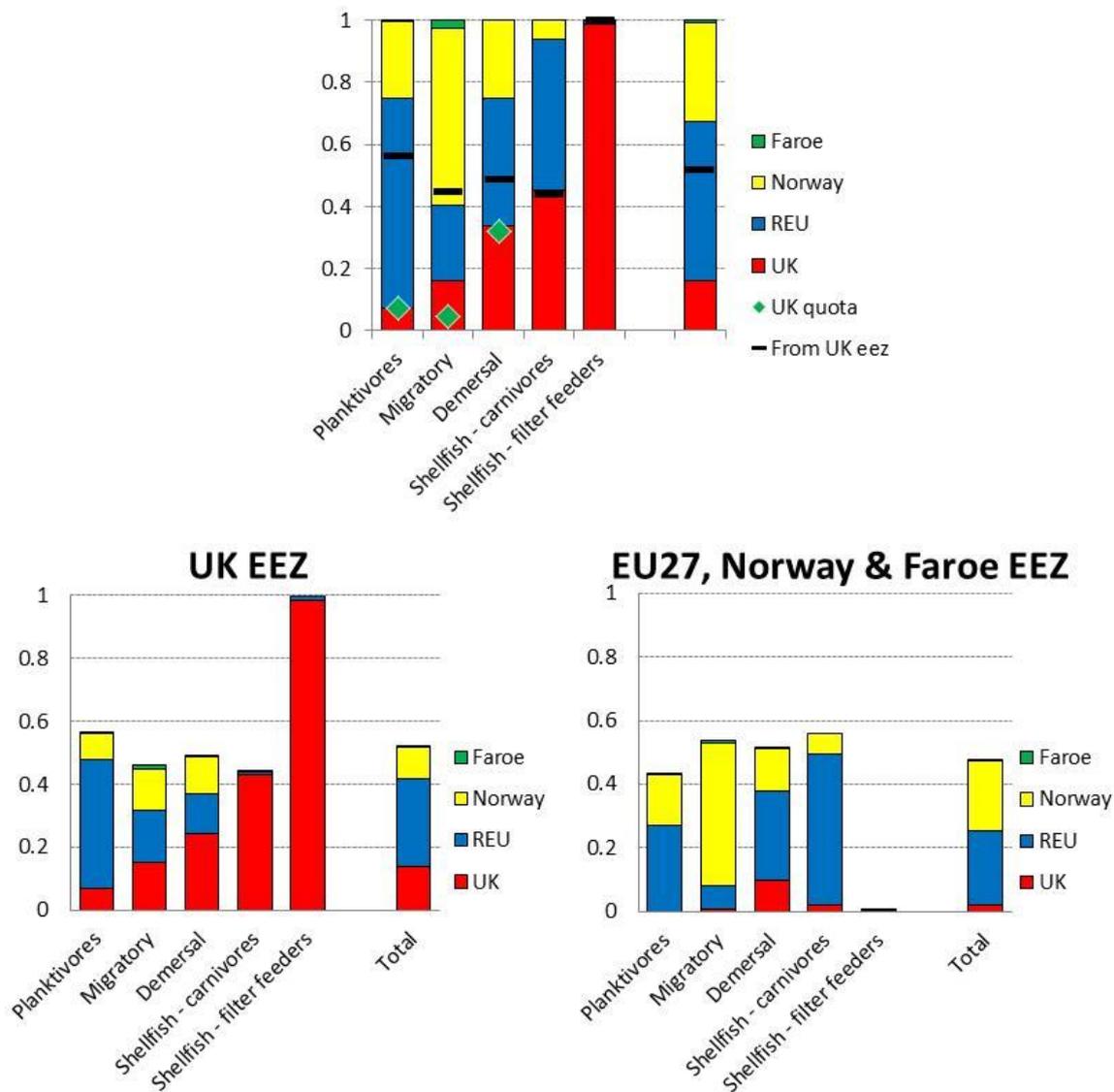


Figure 4. Average distribution of landings between nation-groups and EEZ's for the North Sea during 2003-2013. Top panel: Proportional distributions of landings from the North Sea between the UK, rest of the EU (REU), Norway and the Faroe Islands. The UK quota value corresponds to 2018. The horizontal black bar indicates the proportion of landings taken in the UK EEZ. Lower panels: proportional distribution of landings between nation-groups within the UK EEZ (left), and the remaining non-UK EEZs of the North Sea (right).

Herring (Fig.5 and Fig. A2.11).

The UK's case for negotiating a greater share of the North Sea herring TAC as an independent nation state is very clear from the analysis. At around 0.13, the UK's share of the TAC is considerably less than either the proportion of landings coming from UK waters (0.73), or the zonal allocation of biomass according to the University of Aberdeen (0.88). A significant fraction of the international landings of North Sea herring are taken by Norway (0.37).

Sandeel (Fig.5 and Fig. A2.12).

At their peak, landings of sandeels for fish meal processing matched those of herring in the North Sea, though since 2015 they have been somewhat smaller (Fig. A2.12). The UK has never had a significant stake in this fishery for a variety of reasons, and indeed closed areas of its waters to sandeel fishing in the late 1990's to protect breeding colonies for various seabird species. Nevertheless, 55% of the international sandeel landings during 2003-2013 were taken in the UK EEZ, including a share of the TAC allocated to Norway. The UK does have a small Relative Stability quota for sandeels but regularly trades this with other EU-27 nations in return for quota in other species. One might imagine that providing access to sandeel quota in UK waters post-Brexit could be a significant bargaining point in the negotiations.

Horse mackerel, sprat, blue whiting, Norway pout. (Fig 5)

The UK has almost no quota for sprat or Norway pout in the North Sea. The latter is not caught for human consumption, but for reduction to meal and oil to service livestock rearing (especially in Denmark) and aquaculture (in Norway). In the case of sprat, the fishery is mainly conducted in the EEZ of other EU states, hence the lack of a UK stake in the fishery.

Similarly, the quota for horse mackerel in the North Sea is smaller than the proportion of landings taken from UK waters, though the disparity is not so extreme as for herring and mackerel. The blue whiting TAC is set at the scale of the NE Atlantic since, like mackerel, the species is highly migratory. Only a small proportion is taken in the North Sea (mostly outside the UK EEZ); most of the fishery is conducted in deep water off the shelf edge to the west of the UK. Both Norway and the Faroe Islands have significant interests in the blue whiting fishery.

Mackerel (Fig.5 and Fig. A2.13).

The presence of mackerel in the North Sea (FAO area 27.4), and its distribution between UK, EU27 and Norwegian EEZs, is highly variable from year to year depending in the details of the annual migration route taken by the fish as they travel southwards from the feeding areas in the Norwegian Sea and around Iceland in the autumn and winter, heading for their spawning grounds west of Ireland. Hence, it is difficult to draw strong conclusions on the zonal attachment or EEZ distribution of landings. The quota system for mackerel is complicated since the TAC is set at the scale of the whole NE Atlantic, but a portion of this is assigned to the North Sea and Norwegian Sea regions. Of this, the UK has only a very small share which is not at all commensurate with the proportion of North Sea landings taken in the UK EEZ. However, the UK quota in the North Sea is usually topped up by transfer from the UK's allocation in the waters west of the UK depending on the migration route of the fish.

Norway and to a lesser extent the Faroe Islands also have significant quota allocations for mackerel.

Megrim (Fig.6)

96% of megrim landings in the North Sea were caught in the UK EEZ during 2003-2013 and this corresponded almost exactly with the UK's Relative Stability quota.

Cod (Fig.6 and Fig. A2.14)

The cod fishery in the North Sea has undergone a steep decline since the early 1980's, partly due to warming temperatures and their effect on recruitment, partly due to a northerly shift in distribution of the stock also related to warming, and partly as a legacy of gross over-fishing in the years leading up to the institution of the CFP. Since 1983 the UK share of the international landings has been maintained at a level above the Relative Stability allocation (0.39) by inward trading of quota. Between 2003 and 2013 the realised UK share of landings almost exactly matched the proportion taken from UK waters, though during 2003-2013 this was around 20% less than the zonal allocation estimate from the University of Aberdeen.

Haddock (Fig.6 and Fig. A2.15)

The history of haddock landings from the North Sea shows a massive peak in the late 1970's, partly due to Russia and Norway, in response to a short period of extraordinarily strong recruitments to the stock which presumably had environmental origins though the exact causes are unknown. The UK's Relative Stability quota for haddock is 0.65, while the average proportion of landings taken from the UK EEZ has been higher than this at around 0.7 since 2003, and the zonal allocation according to the University of Aberdeen has been higher still at 0.8. However, the UK share of international landings (including Norway) has exceeded all of these at around 0.9 since the implementation of the CFP, maintained at this level by quota swaps with other nations. Hence, one might expect the haddock allocation to be a particularly difficult issue in any negotiations.

Whiting (Fig.6 and Fig. A2.16)

As for haddock, the UK's share of the international landings has been maintained above the Relative Stability quota through quota swaps during most of the time since 1983, though during 2003-2013 the two were closely aligned according to the STECF data. The proportion of whiting landings taken from the UK EEZ was very high during 2003-2013 (0.9), and exceeded the zonal attachment estimate from Aberdeen University (0.79).

Saithe (Fig.6 and Fig. A2.17)

The UK's Relative Stability share of the TAC for saithe is notoriously low (0.08), which is much smaller than the proportion of landings from the UK EEZ (around 0.51 during 2003-2013), or the two zonal attachment estimates available for this species from the University of Aberdeen (0.46) and the White Paper (0.37).

Anglerfish (Fig.6 and Fig. A2.18)

The UK share of the total landings of anglerfish from the North Sea has been relatively stable at around 0.75 since the 1960's, which corresponds with the Relative Stability quota and the proportion of landings from the UK EEZ. Here we have two estimates of zonal attachment based on biomass distribution but they differ considerably, from 0.61 (University of Aberdeen) to 0.95 (White Paper).

Hake (Fig.6 and Fig. A2.19)

The abundance of hake in the northern North Sea has increased dramatically since 2000 due to a northerly shift in the geographic distribution of the species up the western side of the British Isles. The problems that this has presented to the UK fishing industry are well documented – with a Relative Stability quota share of 0.18 the increased abundance relative to other species has meant that hake has become a so-called 'choke species' under the Landing Obligation. Quota trades between areas within the UK allocation and between nations have enabled the UK quota in the North Sea to increase to around 0.4, in line with the proportion of landings coming from UK waters, but this remains less than the University of Aberdeen zonal attachment estimate of 0.79 (during the period 2011-2015).

Plaice (Fig.6 and Fig. A2.20)

The UK share of the total landings of plaice has remained steady at around 0.27 since the 1980's, which corresponds to both the Relative Stability quota and the proportion of landings taken in UK waters.

Common Sole (Fig.6 and Fig. A2.21)

The UK share of common sole landings from the North Sea has remained steady around 0.04-0.05, consistent with the Relative Stability quota. However, this is substantially smaller than either the proportion of landings from the UK EEZ (around 0.33 during 2003-2013), or the zonal attachment estimate from the White Paper (0.47).

Norway lobster (Fig.7 and Fig. A2.22)

The UK has a Relative Stability quota of 0.84 for Norway lobster, and this coincides with the average proportion of landings in reality, and the proportion of total landings taken from UK waters.

Edible crab, common shrimp and Atlantic scallop (Fig.7)

Landings of crab, shrimp and scallops are largely confined to the inshore waters of member states and therefore not subject to Relative Stability quotas. Hence, the common shrimp fishery is almost entirely carried out beyond the limits of the UK EEZ, and the UK takes only a small fraction of the North Sea landings. The reverse applies for crab and scallop, where the UK takes the majority of landings, and these come mostly from the UK EEZ.

2.2. Summary of findings on individual species allocations in the North Sea

The analysis here reveals that the allocation of North Sea fish and shellfish quota to the UK is much more complicated than the headline infamous statistic in the White Paper would suggest. It is true that other EU states take far (12-times) more fish from UK North Sea waters than the UK takes from non-UK parts of the North Sea. But, 32% (457,000 tonnes) of the total landings from the North Sea are of sprat, sandeel and Norway pout, which are of low value per tonne and in which the UK has had little or no commercial interest in the past, while 44% of these landings are taken from the UK EEZ. Added to this, North Sea herring is highly aggregated during the period of the year when it is targeted by the fishing fleets, and 73% of the annual landings are taken from the UK EEZ, of which the UK has a quota of only 13%. Taken together, these few planktivorous fish species, plus the complicated distribution and quota arrangements for the migratory mackerel and blue whiting, have a disproportionate effect on the ‘all species combined’ infamous statistics.

In general, the UK demersal fish quotas, estimates of zonal attachment, and proportions of landings taken from the UK EEZ, are more closely aligned than for the migratory and planktivorous species. This is especially the case when quota swaps and trades are taken into account. In the case of haddock, quota swaps and trades have resulted in a greater proportion of landings being assigned to the UK than would seem to be justified by either zonal attachment or distribution of landings. Nevertheless, there are three stand-out demersal species for which the UK has a substantially lower Relative Stability share of the TAC than zonal attachments or proportional spatial distributions of landings – these are hake, saithe and common sole. The Relative Stability quotas for cod and whiting also appear low compared to the recent (2003-2013) distributions of landings and estimates of zonal attachment though the discrepancies are less extreme.

Of the shellfish species, only Norway lobster is covered by the Relative Stability rules, and in this case the quota seems entirely reasonable in the absence of any zonal attachment estimates.

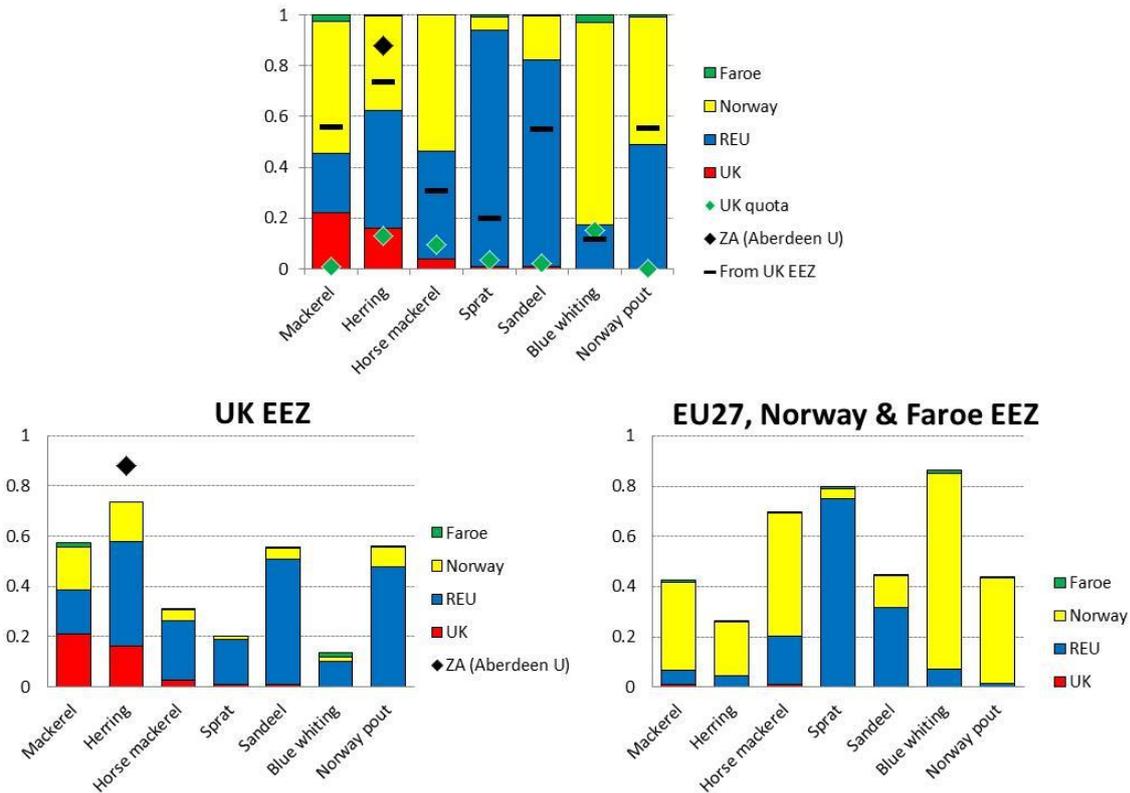


Figure 5. Average distribution of landings of the major planktivorous and migratory fish species between nation-groups and EEZ's for the North Sea during 2003-2013. Top panel: Proportional distributions of landings from the North Sea between the UK, rest of the EU (REU), Norway and the Faroe Islands. The UK quota value corresponds to 2018, while the zonal attachment (ZA) from Aberdeen University corresponds to the period 2011-2015. The horizontal black bar indicates the proportion of landings taken in the UK EEZ. Lower panels: proportional distribution of total North Sea landings for each species between nation-groups within the UK EEZ, together with zonal attachment estimates where available (left), and the remaining non-UK EEZs of the North Sea (right).

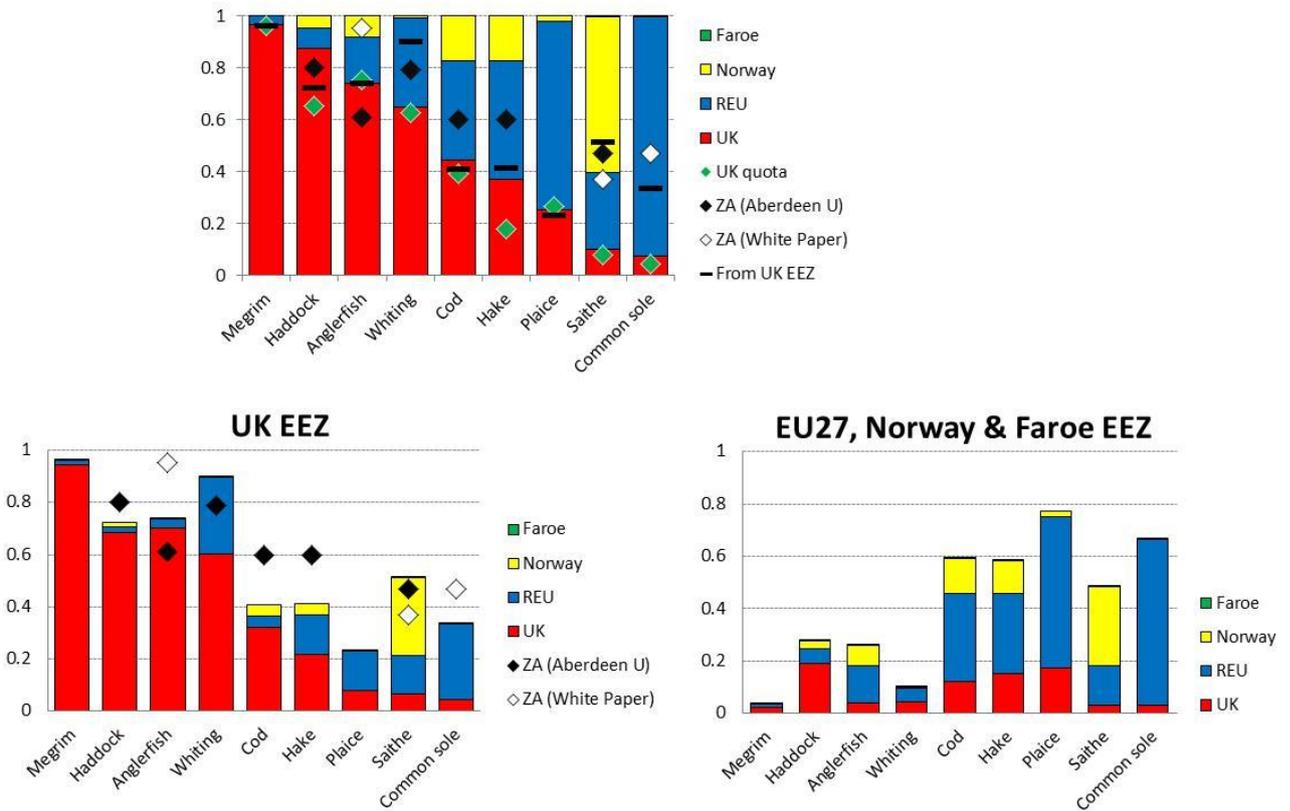


Figure 6. Average distribution of landings of the major demersal fish species between nation-groups and EEZ's for the North Sea during 2003-2013. Top panel: Proportional distributions of landings from the North Sea between the UK, rest of the EU (REU), Norway and the Faroe Islands. The UK quota value corresponds to 2018. The zonal attachment (ZA) from Aberdeen University corresponds to the period 2011-2015, while no dates are given for the swept area zonal attachment estimates in the White Paper. The horizontal black bar indicates the proportion of landings taken in the UK EEZ. Lower panels: proportional distribution of total North Sea landings for each species between nation-groups within the UK EEZ, together with zonal attachment estimates where available (left), and the remaining non-UK EEZs of the North Sea (right).

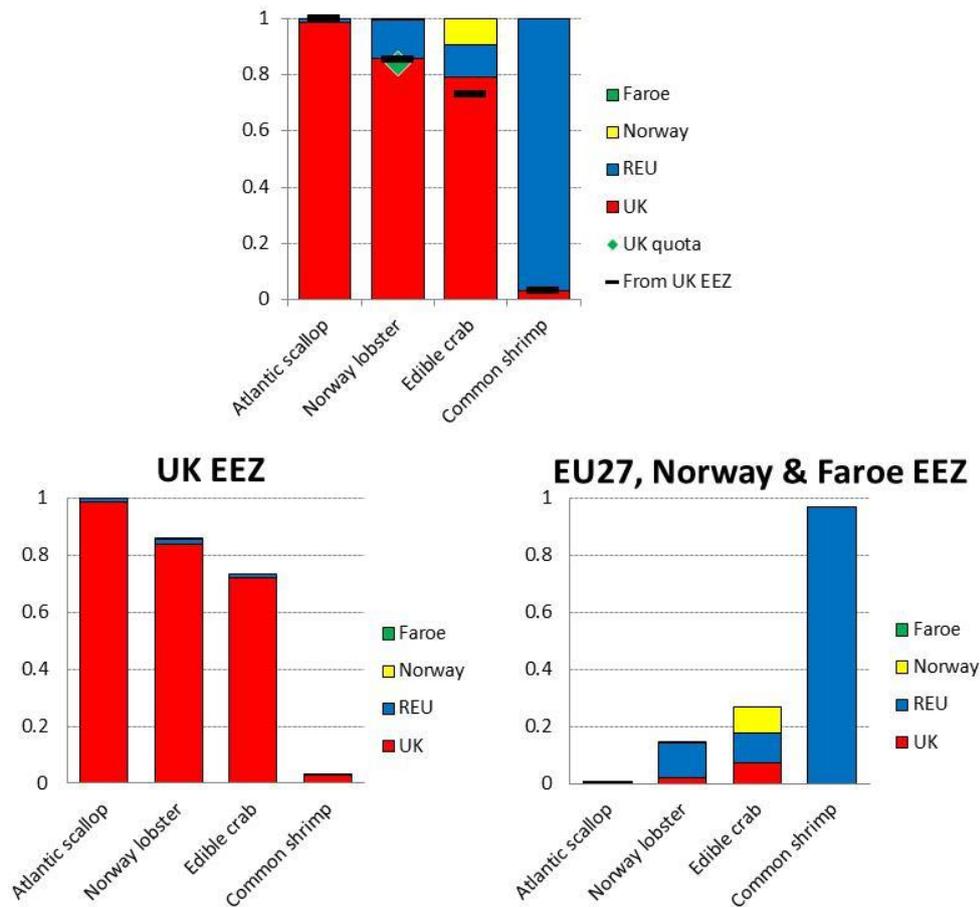


Figure 7. Average distribution of landings of the major shellfish species between nation-groups and EEZ's for the North Sea during 2003-2013. Top panel: Proportional distributions of landings from the North Sea between the UK, rest of the EU (REU), Norway and the Faroe Islands. The UK quota value corresponds to 2018. No estimates of zonal attachment are available for shellfish species. The horizontal black bar indicates the proportion of landings taken in the UK EEZ. Lower panels: proportional distribution of total North Sea landings for each species between nation-groups within the UK EEZ (left), and the remaining non-UK EEZs of the North Sea (right).

2.3. Distribution of fishing gear activity between EEZs in the North Sea.

While the UK remains part of the EU, or potentially during any transition phase, vessels from other EU member states have the right to harvest some or all of their Relative Stability share of each stock's TAC (plus or minus any quota trades) from UK territorial waters under the principle of shared waters which underpins the CFP. Similarly, Norway and the Faroe Islands have reciprocity agreements with the EU which mean that they can also operate in UK waters subject to various prescribed restrictions. When the UK becomes an independent state all these arrangements will cease and reciprocal access will need to be negotiated. Hence, an important part of our project on scoping worst-case scenarios was to determine, not only the

proportion of landings that have been taken from the UK EEZ, but also the proportion of effort that has been expended. Denial of access for non-UK vessels to the UK EEZ is the default sanction available in the absence of any agreement on revised quotas for shared stocks.

The STECF effort data contain records of the hours fished per year for a range of gear classifications, structured by 30' latitude x 60' longitude statistical rectangles, for EU nations only. Although we obtained data on spatially resolved landings by Norwegian vessels to complement the STECF data, we could not access any spatial effort data. Hence we developed a data methodology to up-scale the EU gear effort data to account for Norwegian effort based, essentially, on the ratios of EU species landings to effort per gear (see Appendix 1). Faroese landings, and presumably effort, make up only a very small proportion of the North Sea total and are mainly of mackerel and blue whiting, so in the absence of any spatial data we distributed the Faroese total North Sea landings between UK and non-UK EEZs in proportion to the EU27 fleets, and then applied the method devised for Norwegian data to estimate Faroese effort. We also condensed the STECF gear categories down to 11 to simplify the presentation, and as the basis for using these data in the scenario modelling to follow (see Appendix 1).

The results (Fig. 8) show a strong separation of the UK and other EU-nation's fleets between EEZ waters. UK vessels expend only a small proportion of their effort in non-UK waters. It is difficult to equate effort by one gear to another, but simply summing up the hours spent fishing, the UK accounts for 59% of all the effort in the North Sea and expends only 7% of this in non-UK EEZ waters. The other EU member states together account for 29% of effort and expend 15% of this in the UK EEZ. The UK beam trawl and demersal otter trawl fleets make most use of the non-UK EEZ waters, while the rest-of-the-EU pelagic fleets (pelagic trawl/seine and sandeel/sprat trawl) and demersal beam trawl fleets make most use of the UK EEZ. Norwegian use of UK and non-UK EEZ waters is more evenly distributed across gears. However, the extent to which the rest-of-the-EU pelagic fleets make use of UK EEZ waters seems disproportionately small compared to the disparity in landings. The reason for this is that the landings per unit effort of planktivorous and migratory fish are higher in the UK EEZ than in the non-UK parts of the North Sea (Fig. 9). This implies that the density of fish (biomass per unit area) is higher in the UK zone – which is entirely consistent with survey data for herring and the zonal attachment data. As a consequence, rest-of-the-EU pelagic trawls denied access to the UK EEZ would have to expend 2.9-times more effort to catch the same amount of planktivorous fish (mainly herring) as they do at present. In contrast, landings per unit effort for demersal fish and shellfish are very similar in the UK and non-UK EEZs.

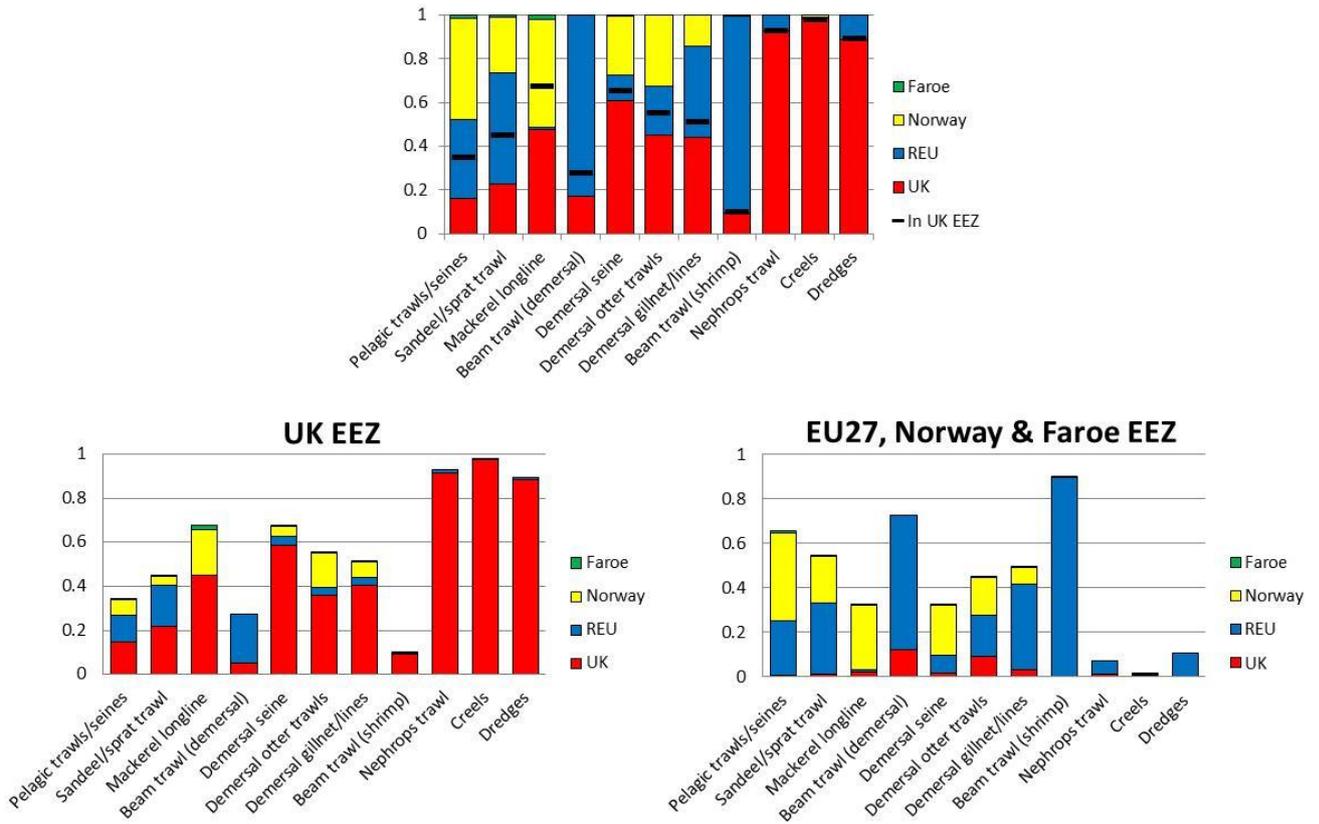


Figure 8. Average distribution of annual effort (hours fished per year) of 11 gear categories between nation-groups and EEZ's for the North Sea during 2003-2013. Top panel: Average proportional distributions of gear effort in the North Sea between the UK, rest of the EU (REU), Norway and the Faroe Islands. Horizontal black bars show the proportion of effort in the UK EEZ. Lower panels: proportional distribution of total North Sea effort for each gear between nation-groups within the UK EEZ (left), and the remaining non-UK EEZs of the North Sea (right).

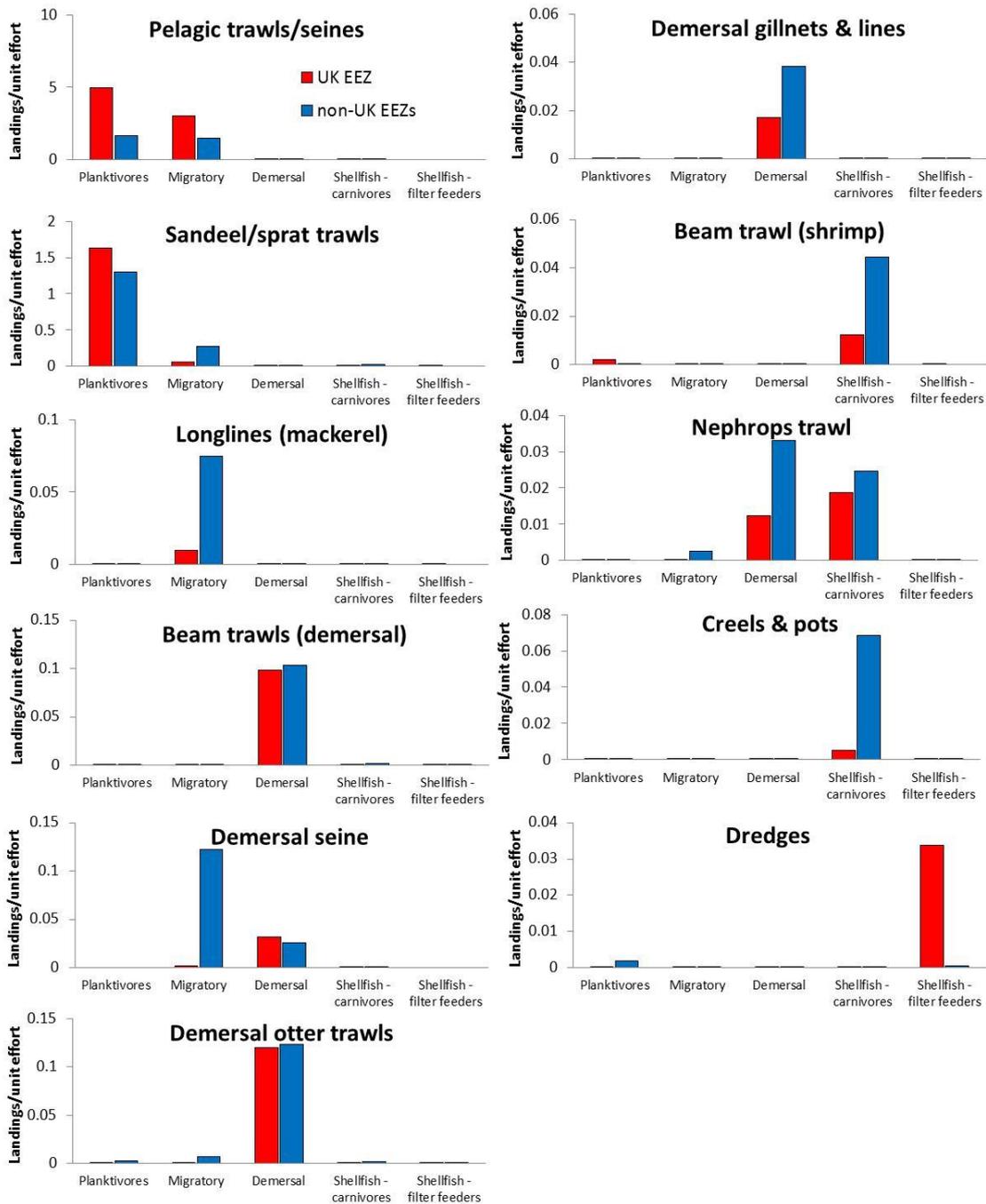


Figure 9. Landings per unit effort (average 2003-2013) for each gear category, in the UK and non-UK EEZs of the North Sea. Data take account of EU, Norwegian and Faroese landings data.

3. Narratives for the post-Brexit unilateralism scenarios

3.1. Quota and effort narratives

We seek to define plausible scenarios of landing quotas and fishing effort that could realistically occur in the event of breakdown of post-Brexit shared-stock negotiations between the EU, UK and Norway and descent into unilateralism. With regard to quotas, our narrative for the sequence of events is as follows:

- The UK goes into negotiations seeking increased share of the TACs for a range of species, guided by estimates of zonal attachment (Figure 10).
- Within the pelagic sector, we assume that the UK is seeking a substantially greater share of the TAC for herring. We expect a similar negotiating goal with regard to quotas for mackerel and horse mackerel but since these stocks are not managed at a scale beyond the confines of the North Sea we cannot address the consequences here in this report.
- We hypothesise that the UK will have no realistic immediate prospect of initiating fisheries for sandeels (or sprats) in its EEZ due to lack of vessels, shore processing infrastructure, and markets.
- Within the demersal sector, we assume that the UK respects existing share of the ICES recommended TAC (\pm quota swaps/trades) for megrim, anglerfish, haddock and plaice, but seeks a greater share of the TAC for cod, whiting, hake, saithe and common sole.
- Remaining EU nations and Norway refuse to yield up quota to the UK despite the threat of denial of access for gears heavily reliant on harvesting fish in the UK EEZ.
- The UK unilaterally awards itself increased quota for herring, cod, saithe and sole.
- The EU and Norway adhere to their existing Relative Stability quota, so the combination of quotas UK, EU and Norwegian fisheries are set to exceed the recommended TACs for herring, cod, saithe and plaice (see Table 1; Figure 11).

With regard to fishing effort, our narrative is that:

- The UK denies access to its EEZ for all foreign vessels. The EU and Norway reciprocate, blocking UK vessels from fishing in their waters.
- Within the UK EEZ, the UK increases effort of all gears to match that of the expelled EU and Norwegian fleets, except for the sandeel/sprat trawls. This will be by a combination of repatriation of effort currently expended in non-UK zones, and new effort.
- Within the non-UK EEZ, the EU and Norway accommodate all of the effort repatriated from the UK zone and attempt to maintain their existing Relative Stability share of the ICES recommended TAC. In the case of pelagic trawls/seines and sandeel/sprat trawls this will imply an increase in effort to compensate for the reduced catchability of pelagic and migratory fish in the non-UK EEZ (see Table 2; Figure 11).

3.2. Discarding narratives

The Defra White Paper (Defra 2018a) is explicit in its commitment to elimination of discarding in UK waters. The Landing Obligation which began to be implemented under the CFP in 2016 and will be fully implemented for all species subject to a TAC limit by 1 January 2019, though the likely degree of compliance and enforcement remains unclear.

For our negotiation-breakdown projections we assumed two scenarios of discarding:

1. Technical developments in fishing gears and practices lead to improved selectivity and the elimination of undersize fish from catches, while quota uplift allows currently over-quota catch to be landed legally. This scenario represents the ambition of the Landing Obligation, and we refer to it as ‘improved selectivity’.
2. Quota-uplift leads to elimination of the over-quota discarding that was occurring prior to the Landing Obligation, but catches of undersize fish continue unchecked and these are discarded. This is perhaps a more realistic scenario, and we refer to it as ‘only-undersize discarding’.

Changes in quotas for North Sea fish species claimed by the UK

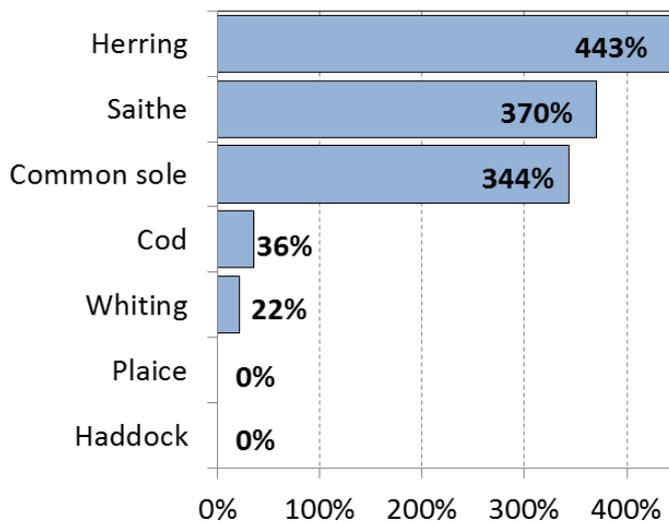


Figure 10. Percentage changes in quota for key fish species in the North Sea which might be plausibly sought by the UK in post-Brexit fisheries negotiations. Values are based on estimates of zonal attachment and pre-Brexit Relative Stability shares adjusted by quota trades and swaps. In the case of haddock, zonal attachment leaves the UK with an 8% deficit of quota share after taking account of quota trades and swaps, so we assume that the UK defends status-quo on haddock. In the case of plaice, there are no zonal attachment data available, but the proportion of total landings taken in the UK EEZ is close to the UK's Relative Stability share so we assume that the UK defends status quo for plaice too.

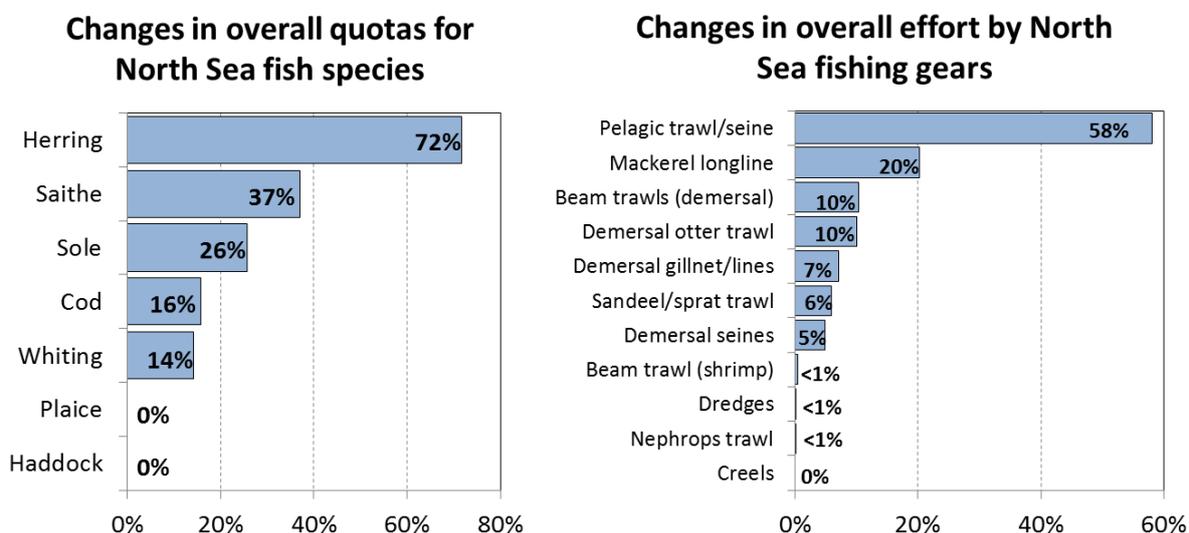


Figure 11. Percentage changes in overall quota for key fish species, and overall effort by fishing gears in the North Sea as a result of a unilateralist outcome from post-Brexit fisheries negotiations. Derivation of values is shown in Tables 1 & 2. Left panel shows the change in the combined quotas of all fishing nations in the North Sea as a result of unilateralism. The increases are a result of the UK claiming quota increases (Figure 10) and the remaining nations adhering to their Relative Stability shares of the TAC. Right panel shows changes in overall fishing effort in the North Sea by different gear groups as a result the UK denying access to the UK EEZ for EU and Norwegian vessels, with reciprocal action against UK vessels. The figures include an element for increased effort required by EU and Norwegian vessels to take their Relative Stability quotas in their own waters where catch per unit effort is lower than in the UK EEZ.

Table 1. Scenarios for changes in fishing mortality rates of key species in the North Sea arising from a worst-case scenario of quota negotiation breakdown post-Brexit.

| Species | Existing UK Relative Stability quota | 2003-2016 average UK share of landings after quota swaps/trades | UK negotiating target ⁽¹⁾ | Self-proclaimed percentage increase in UK share post-breakdown of negotiations | Percentage increase in North Sea-wide fishing mortality relative to status-quo advice, assuming EU and Norway comply with existing Relative Stability quotas |
|----------------|---|--|---|---|---|
| Herring | 0.13 | 0.162 | 0.88* | +443% | +71.8% |
| Cod | 0.389 | 0.442 | 0.60* | +36% | +15.8% |
| Haddock | 0.654 | 0.875 | 0.80* | -8.6% | 0% ⁽²⁾ |
| Whiting | 0.626 | 0.647 | 0.79* | +22% | +14.3% |
| Saithe | 0.080 | 0.100 | 0.47* | +370% | +37.0% |
| Plaice | 0.265 | 0.253 | 0.265*** | 0% | 0% |
| Common sole | 0.043 | 0.075 | 0.333** | +344% | +25.8% |

⁽¹⁾ The negotiating target is assumed to be the biomass zonal attachment where available (indicated by *), or in the absence of zonal attachment the greater of the existing proportional of international landings being taken from the UK EEZ (**) or Relative Stability (***).

⁽²⁾ In the case of haddock, the existing level of UK landings in excess of Relative Stability is sustained by quota swaps (within the UK) and quota trades (with other countries e.g. Norway). It is not clear whether these trades are for fish in the North Sea, or elsewhere in the NE Atlantic – e.g. cod in the Arctic. We assume that the EU and Norway will rescind their quota trades and absorb the gain into their quota for haddock in the North Sea.

Table 2. Scenarios for changes in effort (hours per year) relative to 2003-2013, of different fishing gears in the North Sea arising from a worst-case scenario of quota negotiation breakdown post-Brexit.

| Gear | 2003-2013 total international effort in the North Sea, including Norway and Faroe (h.y⁻¹) | 2003-2013 total effort in the UK EEZ (h.y⁻¹) | 2003-2013 non-UK effort in the UK EEZ (h.y⁻¹) | 2003-2013 UK effort in the UK EEZ (h.y⁻¹) | 2003-2013 UK effort outside the UK EEZ (h.y⁻¹) | Additional UK effort post-breakdown (h.y⁻¹) ⁽¹⁾ | Additional EU, Norway & Faroe effort post-breakdown (h.y⁻¹) ⁽²⁾ | Percentage change in total effort post-breakdown |
|------------------------|---|--|---|---|--|--|--|---|
| Pelagic trawl/seine | 142580 | 49428 | 28061 | 21366 | 1365 | 26696 | 56124 | +58.1% |
| Sandeel/sprat trawl | 253029 | 114563 | 59400 | 55163 | 2836 | 0 | 14850 | +5.9% |
| Mackerel longline | 87988 | 59501 | 19816 | 39685 | 2085 | 17731 | 0 | +20.2% |
| Beam trawls (demersal) | 569096 | 157199 | 126971 | 30228 | 68244 | 58727 | 0 | +10.3% |
| Demersal seines | 1005 | 680 | 87 | 593 | 17 | 47 | 0 | +4.8% |
| Demersal otter trawl | 1552308 | 858987 | 296628 | 562359 | 141307 | 155320 | 0 | +10.0% |
| Demersal gillnet/lines | 411291 | 209750 | 42916 | 166834 | 14174 | 28742 | 0 | +7.0% |
| Beam trawl (shrimp) | 592280 | 58923 | 2565 | 56357 | 421 | 2144 | 0 | +0.4% |
| Nephrops trawl | 875980 | 814120 | 10732 | 803388 | 10222 | 509 | 0 | +0.01% |
| Creels | 1582042 | 1551537 | 7693 | 1543843 | 13044 | 0 | 0 | 0 |
| Dredges | 166215 | 148785 | 1698 | 147087 | 467 | 123 | 0 | +0.07% |

⁽¹⁾ Additional UK effort arises from expansion of existing fleet activities (including those repatriated from operating in non-UK EEZ waters) to fill the void created by the expulsion of EU and Norwegian vessels from the UK EEZ. We assume that the UK makes no attempt in the short-medium term to expand sandeel/sprat trawl effort for a variety of reasons as explained in the text.

⁽²⁾ Additional non-UK effort arises from EU, Norwegian and Faroese attempts at maintaining their status-quo catches of pelagic and migratory fish from the EU and Norwegian EEZs where catch rates per unit effort are lower. We assume UK EEZ catch per unit effort of pelagic trawls is 3-times that in non-UK EEZ waters, and 1.25-times for sandeel/sprat trawls. All other gears have the same catch rate per unit effort in UK and non-UK waters.

4. Modelling the consequences of unilateralism scenarios

4.1. Projections based on single species modelling

4.1.1. Introduction

The majority of ICES stock assessments are based on models of single species population dynamics that do not consider biological interactions between species or other environmental effects [75]. Such models are likely to perform adequately in the estimation of population size and mortality rates given historical observations of catches and research vessel surveys of relative abundance. Such models will also perform adequately in making short term forecasts of abundance provided the projected populations include cohorts of fish that have been measured in the baseline population from which the projection is made. Longer term projections beyond about five years are much more speculative because they assume constancy in biological parameters such as growth, maturity and natural (non-fishing) mortality. Crucially they also depend upon a relationship between spawning stock size (SSB) and subsequent recruitment which is usually difficult to model realistically. Nevertheless such models are widely used in the determination of biological reference points which attempt to quantify equilibrium conditions in the long term.

Despite the limitations of single species models they have the advantage of requiring fewer parameters to estimate and they can be more robust than complex models with highly non-linear relationships between interacting species. We therefore ran analyses based in single species models in order to provide some insight into the possible effects of increased fishing mortality in a post-Brexit scenario. The demersal species are higher up the food chain and will be less influenced by predatory interactions than herring and hence the projections may be less biased by ignoring predatory interactions.

A full description of the single species projection modelling is provided in Appendix 3

4.1.2. Results

The changes projected under the post-Brexit scenario are summarised in Table 3 and Figures 12 & 13 (short term; 5 year horizon), and Table 4 and Figure 14 & 15 (long term, >20 years) for the seven species considered (herring, cod, haddock, whiting, saithe, plaice and sole). As is expected the higher fishing mortality assumed under the post-Brexit unilateralism scenario results in lower biomass both in the short and long term. The largest reductions are seen for herring, cod, saithe and sole. For most species the change in landings is negligible but herring landings show an increase. The most noteworthy changes are the large increase in the risk of the biomass falling below the precautionary biomass threshold in herring and cod. At current rates of fishing, both these stocks have around a 30% chance of falling below Bpa (Table 3). The post-Brexit scenario more or less doubles this risk. For sole there is a very large increase in the relative risk (final column in Tables 5 & 6). However, this is a multiple of a very low

probability (0.4) and the overall probability of falling below Bpa in the post-Brexit case remains low.

Sensitivity model runs with Ricker recruitment, and using different assumptions about recruitment, show that results can change markedly for the probability of falling below Bpa (Tables 5 & 6), though the short term effects are somewhat less sensitive to the recruitment assumption. In the case of herring a ‘Ricker recruitment model’ assumption rather than the standard ‘Beverton-Holt model’ used by ICES, leads to much larger reductions in projected biomass and negative effects on landings in the long term.

Table 3. Short term changes (year 5) to SSB, landings and the probability of the biomass falling below Bpa.

| Species | % change in SSB | % change in landings | % prob. SSB<Bpa at Fsq | % prob. SSB<Bpa with post-Brexit scenario | Relative increase in prob. SSB<Bpa |
|---------|-----------------|----------------------|------------------------|---|------------------------------------|
| Herring | -22.76 | 17.18 | 34 | 76.5 | 2.249 |
| Cod | -10.94 | -2.96 | 59 | 73.2 | 1.240 |
| Haddock | 0 | 0.0 | 6.8 | 6.8 | 1.000 |
| Whiting | -1.86 | 8.23 | 0 | 0 | NA |
| Saithe | -23.58 | -0.18 | 0 | 0 | NA |
| Plaice | 0 | 0 | 0 | 0 | NA |
| Sole | -14.78 | 0.73 | 0.4 | 18.5 | 46.333 |

Table 4. Long term changes (at equilibrium) to SSB, landings and the probability of the biomass falling below Bpa.

| Species | % change in SSB | % change in landings | % prob. SSB<Bpa at Fsq | % prob. SSB<Bpa with post-Brexit scenario | Relative increase in prob. SSB<Bpa |
|---------|-----------------|----------------------|------------------------|---|------------------------------------|
| Herring | -23.80 | 17.34 | 30.7 | 71.7 | 2.337 |
| Cod | -15.37 | -2.71 | 32.1 | 56.3 | 1.753 |
| Haddock | 0.0 | 0.0 | 9.7 | 9.7 | 1.003 |
| Whiting | -2.28 | 8.25 | 0 | 0 | NA |
| Saithe | -28.95 | -0.25 | 0 | 0 | NA |
| Plaice | 0.0 | 0.0 | 0 | 0 | NA |
| Sole | -24.55 | 0.73 | 0.4 | 7.7 | 19.356 |

Table 5. Cod. Comparison of projection results using the standard Beverton-Holt model with Ricker model. The Ricker model uses all the available stock recruitment data. Long term results are highlighted in blue. Short term results are highlighted in gold.

| Recruitment/ Time horizon | % change in SSB | % change in landings | % prob. SSB<Bpa at Fsq | Relative increase in prob. SSB<Bpa |
|------------------------------|--------------------|-------------------------|------------------------------|---|
| Beverton-Holt Long term | -15.37 | -2.71 | 32.1 | 1.753 |
| Ricker Long term | -9.35 | 3.96 | 0.01 | 14.097 |
| Beverton-Holt Short term | -10.94 | -2.96 | 59 | 1.24 |
| Ricker Short term | -12.87 | -3.63 | 11.36 | 1.98 |

Table 6 Herring. Comparison of projection results using the standard Beverton-Holt model with Ricker model. The Ricker model uses all the available stock recruitment data. Long term results are highlighted in blue. Short term results are highlighted in gold.

| Recruitment/ Time horizon | % change in SSB | % change in landings | % prob. SSB<Bpa at Fsq | Relative increase in prob. SSB<Bpa |
|------------------------------|--------------------|-------------------------|------------------------------|---|
| Beverton-Holt Long term | -23.80 | 17.34 | 30.70 | 2.33 |
| Ricker Long term | -44.38 | -14.35 | 13.67 | 5.15 |
| Beverton-Holt Short term | -22.76 | 17.18 | 34 | 2.249 |
| Ricker Short term | -34.36 | 5.099 | 28.13 | 2.77 |

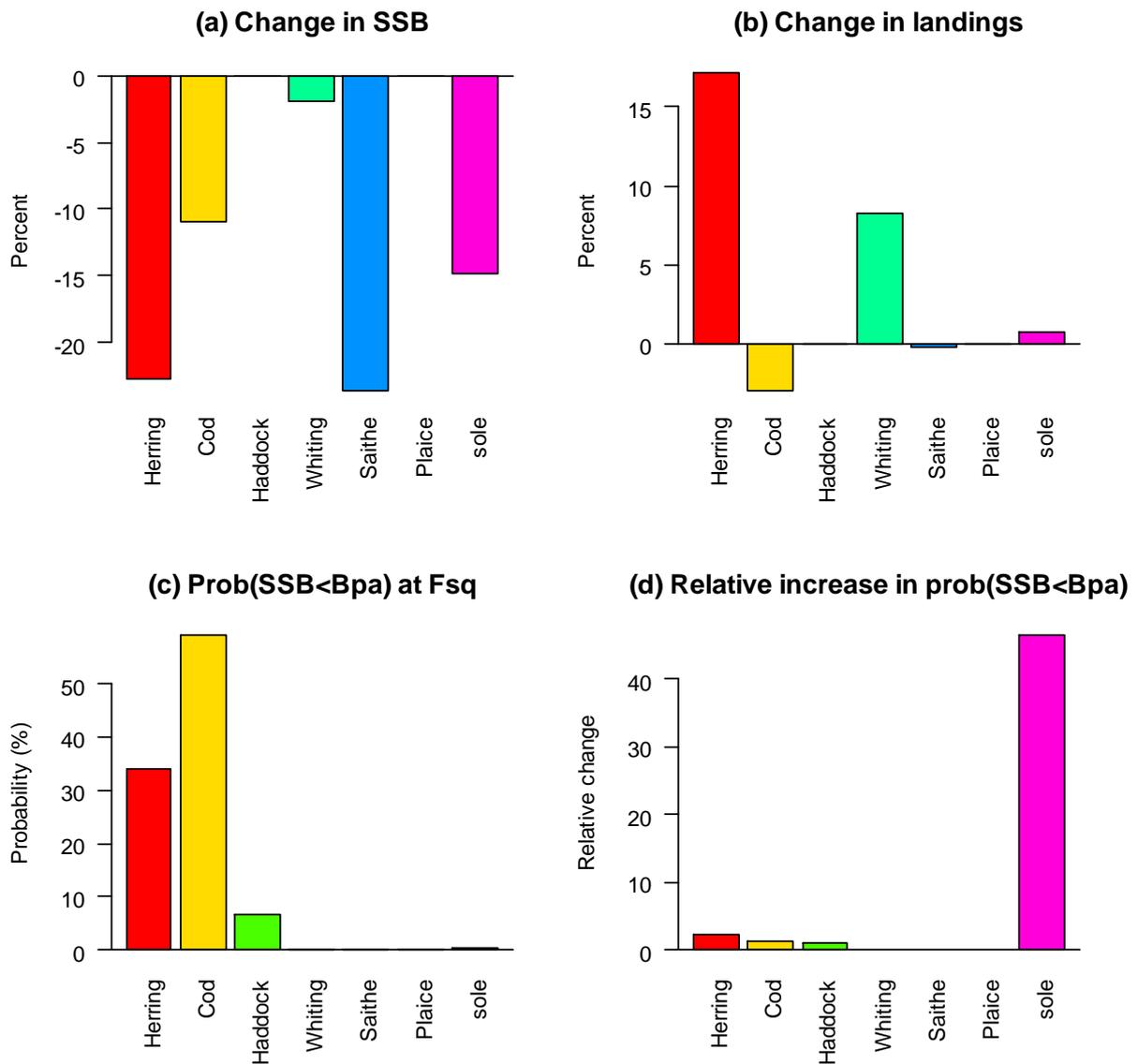


Figure 12. Short term effects (at year 5) of post-Brexit scenarios. (a) the percentage change in the mean SSB in year 5. (b) the percentage change in the mean human consumption landings in year 5. (c) the probability that the biomass will be below Bpa at the current rate of fishing mortality (Fsq). (d) the relative increase in the probability of falling below Bpa in the post-Brexit scenario.

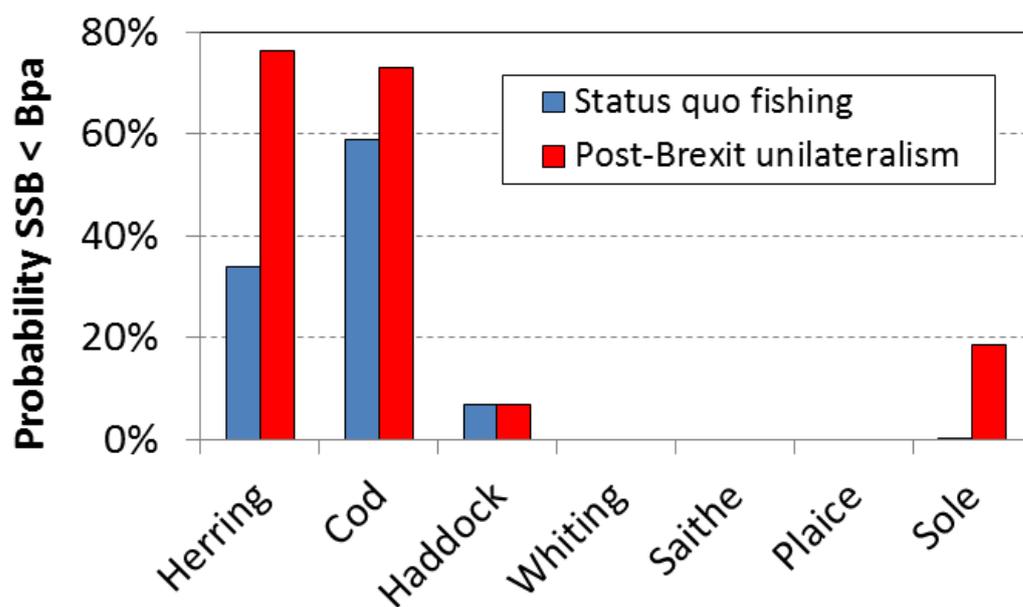


Figure 13. Comparison of short term (5 year) percentage probabilities of the spawning stock biomass (SSB) of key species in the North Sea falling below their respective precautionary levels (Bpa). Blue bars indicate the probabilities assuming that status quo fishing conditions in 2017 continue unchanged into the future. Red bars indicate the probabilities under the post-Brexit unilateralism scenario.

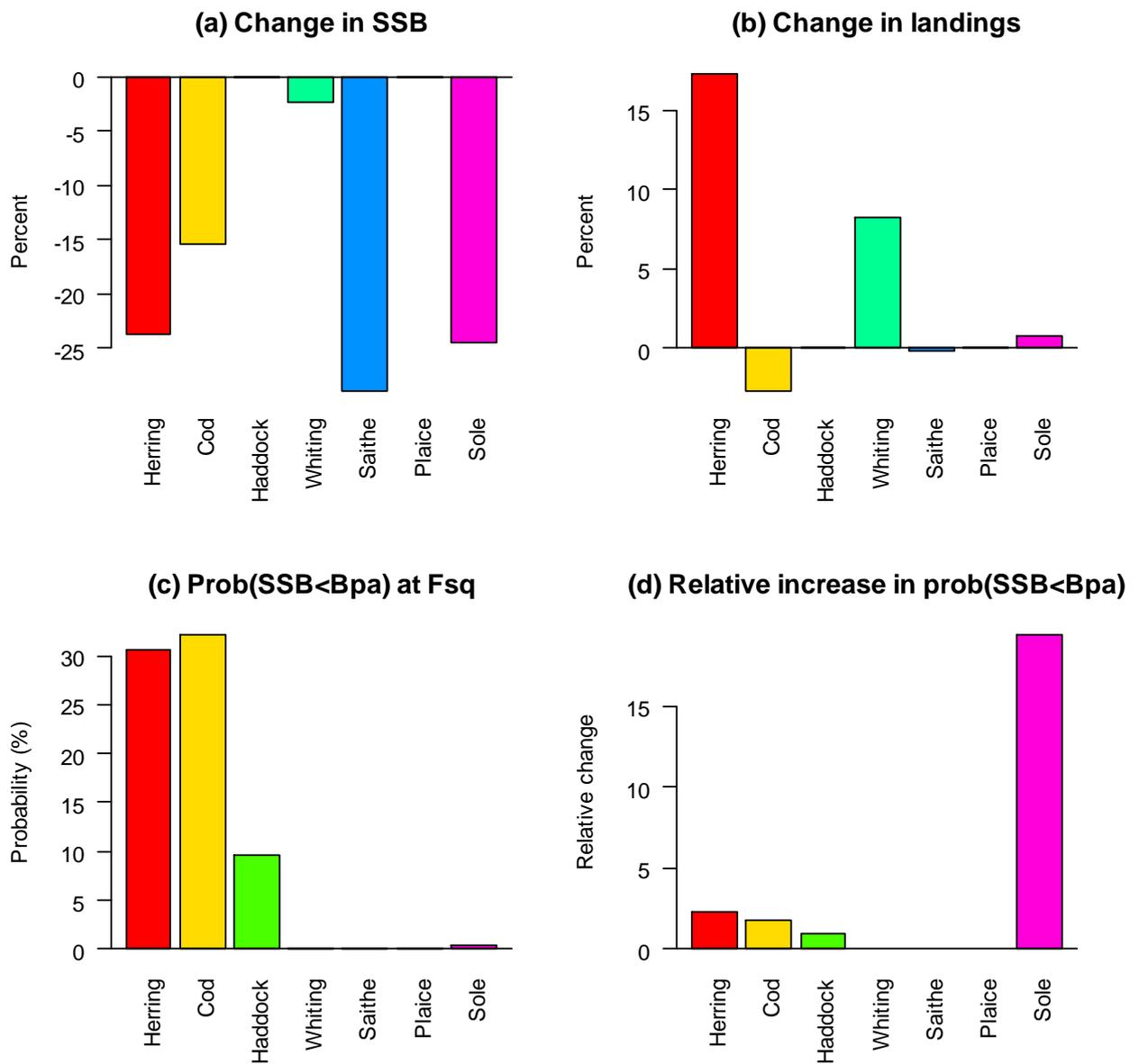


Figure 14. Long term effects of post-Brexit scenarios. (a) the percentage change in the mean equilibrium SSB. (b) the percentage change in the mean equilibrium human consumption landings. (c) the probability that the biomass will be below Bpa at the current rate of fishing mortality (Fsq). (d) the relative increase in the probability of falling below Bpa in the post-Brexit scenario.

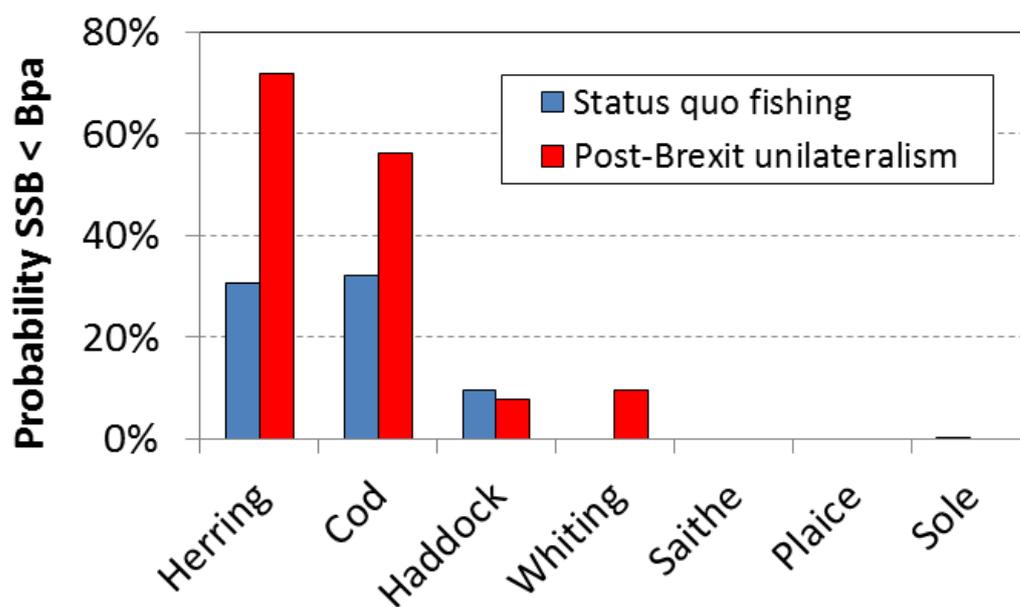


Figure 15. Comparison of short term (5 year) percentage probabilities of the spawning stock biomass (SSB) of key species in the North Sea falling below their respective precautionary levels (Bpa). Blue bars indicate the probabilities assuming that status quo fishing conditions in 2017 continue unchanged into the future. Red bars indicate the probabilities under the post-Brexit unilateralism scenario.

4.1.3. Discussion

The single species analyses suggest that of the seven species examined, the post-Brexit scenario would affect herring and cod significantly. These two stocks have a comparatively high probability of being below the precautionary biomass reference points even at current rates of fishing. Elevated rates of fishing mortality projected under the post-Brexit scenario more than doubles this risk for herring and increases it by 75% for cod. Both stocks have a history of depletion and are known to be vulnerable to collapse [76, 77]. The risks are even higher in the short term, especially for cod where the current SSB is already below Bpa according to the latest ICES advice [75].

The projected declines in saithe and sole biomass are larger than those for herring and cod, but as these stocks are already well above Bpa the elevated risk to the biomass in the longer term remains very low. However, it should be noted that these reduced biomasses (by around 25%) are associated with little or no gain in landings and the removal of standing biomass is likely to have ecosystem consequences.

The results from the model used in these single species projections is heavily dependent on the stock-recruitment relationship used. A number of factors are of particular importance. For consistency, we used the year range of recruitment observations adopted for calculating stock reference points by ICES. Typically the years are chosen to reflect what is believed to be the current productivity of the stock (flatfish excluded) and limits the time series to recent years. While this may realistically capture recent stock productivity the very limited number of data points and their inherent variability means fitting a stock recruitment model is extremely unreliable. We incorporated this uncertainty by bootstrapping model parameters and averaging over recruitment models. However, projected populations may lie outside the range of observations used to fit the recruitment model and can generate unrealistic values. For the stocks concerned, the fit of the Beverton-Holt model was asymptotic over the range of observations (Appendix 3) and is therefore likely to be robust unless the stock size is substantially reduced. It may over-estimate likely recruitment at low stock biomass and give an over-optimistic perception of equilibrium biomass.

The sensitivity runs using the Ricker assumption for recruitment in herring and cod gave qualitatively similar results but with a substantial reduction in the risk of falling below Bpa. This arises mainly because the longer time series of data used to fit the model include periods of higher productivity than the standard ICES approach. There are good reasons to believe that recent lower productivity is an environmental effect in both stocks [78, 79] which would support using the shorter time series, and by implication the higher risks of falling below Bpa.

4.2. Projections based on food web modelling

4.2.1. Introduction

The effects of human-induced pressures such as fishing, applied to any part of an ecosystem are eventually felt everywhere to some extent through the phenomenon known as a ‘trophic cascade’ [80]. Cascading effects are attenuated or amplified as they propagate through the food web, depending on the nature of the pressure and details of the ecology. There is no universal theory of trophic cascade propagation [81], so diagnosing the type and magnitude of disturbances that an ecosystem can sustain before being fundamentally altered requires simulation with mathematical models that aim to represent the key ecological components and processes which govern cascades.

We used the StrathE2E marine food web/ecosystem model (version 2.0.0) which was developed at the University of Strathclyde to address questions about the ecosystem effects of fishing [72]. The model spans the entire ecosystem from inorganic and organic nutrients through to birds and mammals but, in order to do so, takes a highly macroscopic view of ecology, aggregating over the many microscopic details of taxonomy, demography and spatial structure (Figure 16). The aim was a scheme which represents the gross dynamics of the ecosystem with a tolerable parameter count, and is fast to run so enabling computational optimisation and sensitivity analysis of model outputs. The scheme comprises a fishing fleet model and an ecology model, with one-way coupling between the two. The fishing fleet model aggregates data on the activity, distributions, and properties of a range of fishing gears over the area covered by the model, and generates the information that is required in the ecology model to simulate the catch, landings, discards, and biomass-dynamics of the targeted and by-catch ecology groups in the sea. Both models build on precedence established by other marine ecosystem models and two earlier prototypes [72, 74].

The data which define the model for a specific geographic region and period of years are grouped into physical oceanographic, ecological, and fishing-related parameters. The physical parameters include bathymetry and seabed sediment types, monthly sea temperatures, river freshwater and nutrient inflows, ocean transport and nutrient flows, atmospheric deposition of nutrient. The ecological parameters include feeding rates and prey preferences of each biological group in the food web, geochemical rates governing the processing of organic matter by bacteria, migration rates for fish and top-predators, and external stocks of migratory fish (mackerel). For the fishing fleet model, the inputs are the activity rates and spatial distributions, selectivity, seabed abrasion, and discarding rates of each of the twelve gear types which are represented. For the North Sea model, these include the 11 groups of STECF gears listed in Table 2, plus Norwegian whaling vessels which operate in the Norwegian sector of the North Sea.

The selectivity parameters of the fishing gears represented in the model are particularly important. These define the range of ecology model groups captured by each gear, and the intensity of fishing mortality inflicted on each group. For example, demersal otter trawls

target demersal fish, but also takes a by-catch of planktivorous fish and benthos, so we need to represent this in the model in order to fully represent the effects of otter trawls on the food web. The selectivity parameters of each gear for fish and benthos groups can be derived from the landings and discards data by gear held in the STECF database. However, certain fishing gears are also known to take an unintended by-catch of top predators (seabirds, seals and cetaceans), and the selectivity parameters for these groups are much more difficult to estimate, but are especially important for a full assessment of the ecosystem impacts of fishing. Details of the derivation of selectivity patterns for top predators are given in Appendix 4.

Most of the ecological parameters are set by computational optimisation to produce a best-fit of the model to a wide range of independent data on the ecological state of the ecosystem during the period defined by the oceanographic and fishing inputs.

The model is available as a package for the R statistical computing environment from the site: <https://gitlab.com/MarineResourceModelling/StrathE2E>. Installation instructions for Windows, Mac-OS and Linus are available from the site. The package contains a user manual and full documentation, as well as the configuration files for the North Sea model used in this report. R-scripts using the StrathE2E2 package to generate the results presented in this report are provided in Appendix 5.

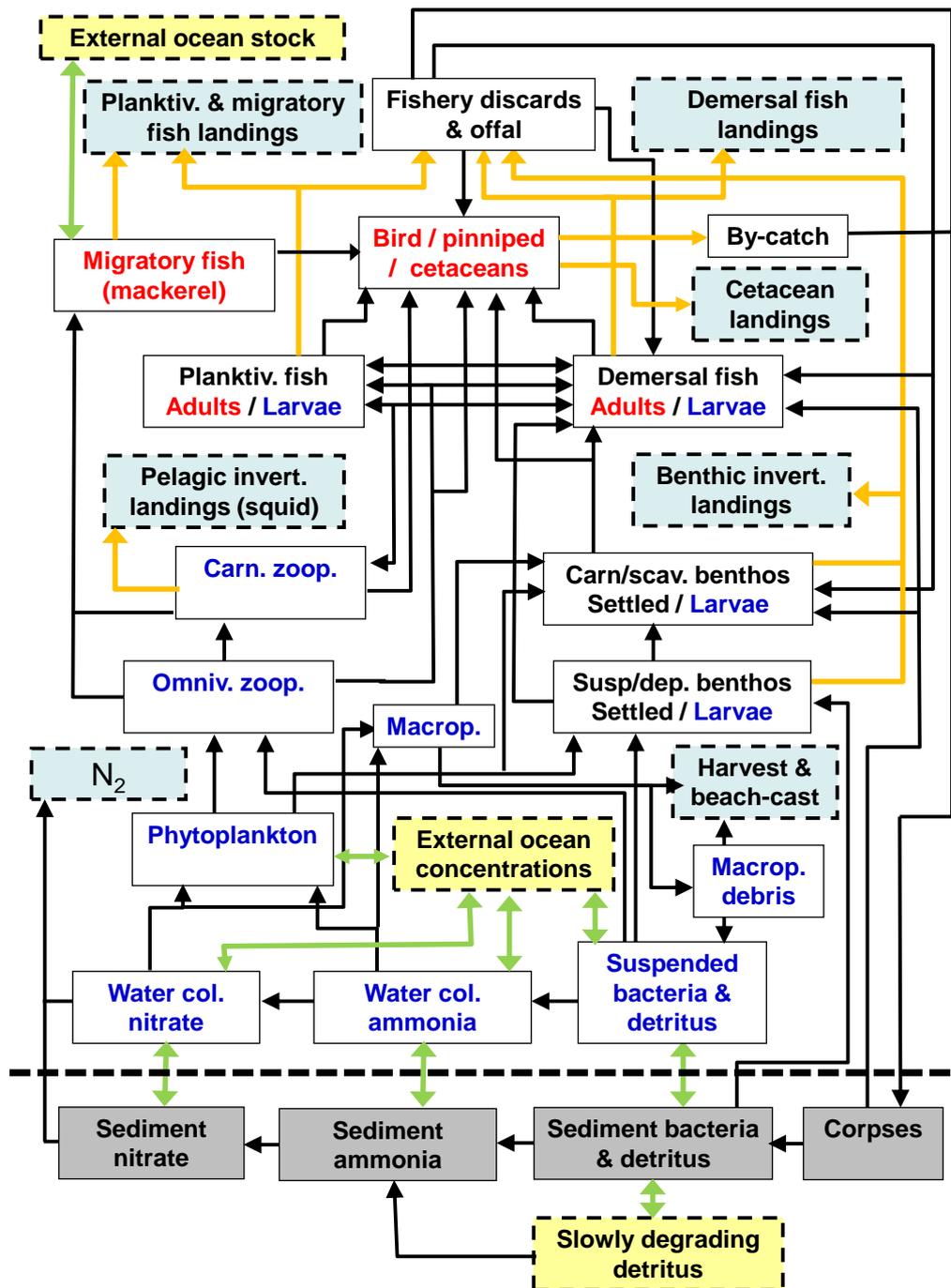


Figure 16. Schematic of the food web compartments of the StrathE2E2 model. Green arrows represent advection, mixing and migration; orange arrows represent fishery-related fluxes; black arrows represent biological fluxes. Red labelled components are active migrators whilst blue are subject to passive advection and mixing and black are anchored. Pale blue boxes represent quantities that are exported from the model whilst yellow are imported. The model also includes fluxes from living components to ammonia, detritus and corpses due to excretion, defecation and death but these are not shown for clarity. Also for clarity, birds, pinnipeds and cetaceans are combined as a single box but in the model are separate entities. The abbreviation “Macrop.” is shorthand for macrophytes.

Baseline model configuration

We configured a model of the North Sea ecosystem for the period 2003-2013 to act as a baseline for the projections of Landing Obligation and post-Brexit unilateralism scenarios. This included 2003-2013 environmental conditions, and fishing and discarding patterns based on the STECF data. The parameters of the ecology model were computed to provide a best-fit to ecological and fishery landings, discards and by-catch data for 2003-2013. Full details of the model parameterisation are provided in Appendix 4.

Post-Brexit unilateralism scenario models

The fundamental change to the baseline model so as to represent the consequences of post-Brexit unilateralism, was the application of percentage changes in the activity rates of the 11 STECF gear groups as defined in Table 2. We assumed no change in the activity rate of Norwegian whalers since these operate completely independently of the CFP or of any fishery negotiations.

We assumed that the seabed abrasion rates of the gears represented in the model were unaffected by the post-Brexit scenario. However we could not necessarily make this assumption with regard to selectivity and discarding of fish and benthos. The STECF data which formed the basis for parameterising discard rates by each gear represent the situation prior to implementation of the Landing Obligation, which was (at least officially) fully implemented by January 2019. The White Paper [3] makes it very clear that the UK will remain committed to preventing the wasteful practice of discarding post-Brexit, though the likelihood of compliance and enforcement remain unclear.

We performed four scenario runs of the model into the future, each starting from the 2003-2013 baseline state (Table 7). The runs reflected combinations of the two discarding scenarios outlined in section 3, and the increments to gear activity rates that might occur in the event of negotiation breakdown (Table 2). Sets of model run were carried out for periods of 2, 5, 10 and 20 years, and results averaged or integrated over the final year in each case.

Table 7. *Scenario combinations of unilateral increments in effort and discarding as the basis for the food web more projections*

| Scenario | Gear activity rates | Discarding conditions |
|----------|---|-----------------------------|
| 1 | As in the baseline model (successful negotiation outcome) | ‘Improved selectivity’ |
| 2 | As in the baseline model (successful negotiation outcome) | ‘Only-undersize discarding’ |
| 3 | Negotiation-breakdown increments in gear activity rates added to the baseline | ‘Improved selectivity’ |
| 4 | Negotiation-breakdown increments in gear activity rates added to the baseline | ‘Only-undersize discarding’ |

We then performed a series of comparisons between model runs to examine the effects of full and partial implementation of the Landing Obligation and its interaction with the post-Brexit negotiation scenarios (Table 8).

Table 8. Combinations of model run comparisons to reveal the interaction between implementation of the Landing Obligation and outcomes of post-Brexit negotiations for the North Sea ecosystem.

| Comparison | Model runs compared | Interpretation |
|------------|--------------------------|---|
| 1 | Baseline vs Scenario 1 | Future projection of the state of the North Sea <u>relative to 2003-2013</u> , assuming full implementation of the Landing Obligation (improved demersal fish selectivity) and continued international adherence to overall catch limits. |
| 2 | Baseline vs Scenario 2 | Future projection of the state of the North Sea <u>relative to 2003-2013</u> , assuming partial implementation of the Landing Obligation (undersize demersal fish discarding continues) and continued international adherence to overall catch limits. |
| 3 | Baseline vs Scenario 3 | Future projection of the state of the North Sea <u>relative to 2003-2013</u> , assuming full implementation of the Landing Obligation (improved demersal fish selectivity) and unilateral actions following breakdown of post-Brexit negotiations. |
| 4 | Baseline vs Scenario 4 | Future projection of the state of the North Sea <u>relative to 2003-2013</u> , assuming partial implementation of the Landing Obligation (undersize demersal fish discarding continues) and unilateral actions following breakdown of post-Brexit negotiations. |
| 5 | Scenario 1 vs Scenario 3 | Future projection of the effects of unilateral actions following Brexit negotiation breakdown assuming full implementation of the Landing Obligation by all nations regardless of Brexit. |
| 6 | Scenario 2 vs Scenario 4 | Future projection of the effects of unilateral actions following post-Brexit negotiation breakdown assuming partial implementation of the Landing Obligation by all nations regardless of Brexit. |

4.2.2. Results

Comparisons 1 and 2: effects of alternative narratives on discarding, given a successful outcome of post-Brexit negotiations.

With a successful negotiation outcome (i.e. the overall international activity rates of gears remains as in the baseline model), the two alternative discarding scenarios have very different outcomes for the ecosystem relative to the 2003-2013 baseline state. The ‘improved selectivity’ scenario which eliminates the capture of undersized demersal fish and discarding, leads to a 5-10% increase in demersal fish biomass. In the long-run this should result in increased TACs and increased quotas. ‘Improved selectivity’ also leads to increases in carnivorous zooplankton, carnivorous/scavenge feeding benthos and seals after 20 years. However, there is also a reduction of 10% in pelagic (planktivorous) fish biomass and smaller reductions in cetacean and seabirds (Figure 17). In the case of ‘only-undersize’ discarding the overall harvesting rates remain unchanged from the baseline so the only effect is a reduction in the food supply for scavenge-feeding taxa in the food web. This leads to small (< 0.4%) reductions in the annual average biomasses of cetaceans, seals, seabirds, demersal fish and carnivorous/scavenge feeding benthos (Figures 18 & 19). The reason for this difference is that ‘improved selectivity’ effectively constitutes a reduction in the harvest ratio (fishing mortality) on demersal fish and their consequent increase in biomass has cascading predator-prey consequences through the food web.

In terms of landings and by-catch, both of the discarding scenarios lead to immediate increases in the landings of demersal fish and reductions in discarding – because in both cases over-quota fish which were being discarded in the 2003-2013 baseline are being landed under the two scenarios (Fig. A4.4 and A4.5). In the ‘improved selectivity’ scenario the changes in the food web caused by the effective reduction in demersal fish harvest ratio eventually (after 20 years) lead to small reductions in pelagic (planktivorous) fish landings, and both the whaling catch and fishery by-catch of cetaceans.

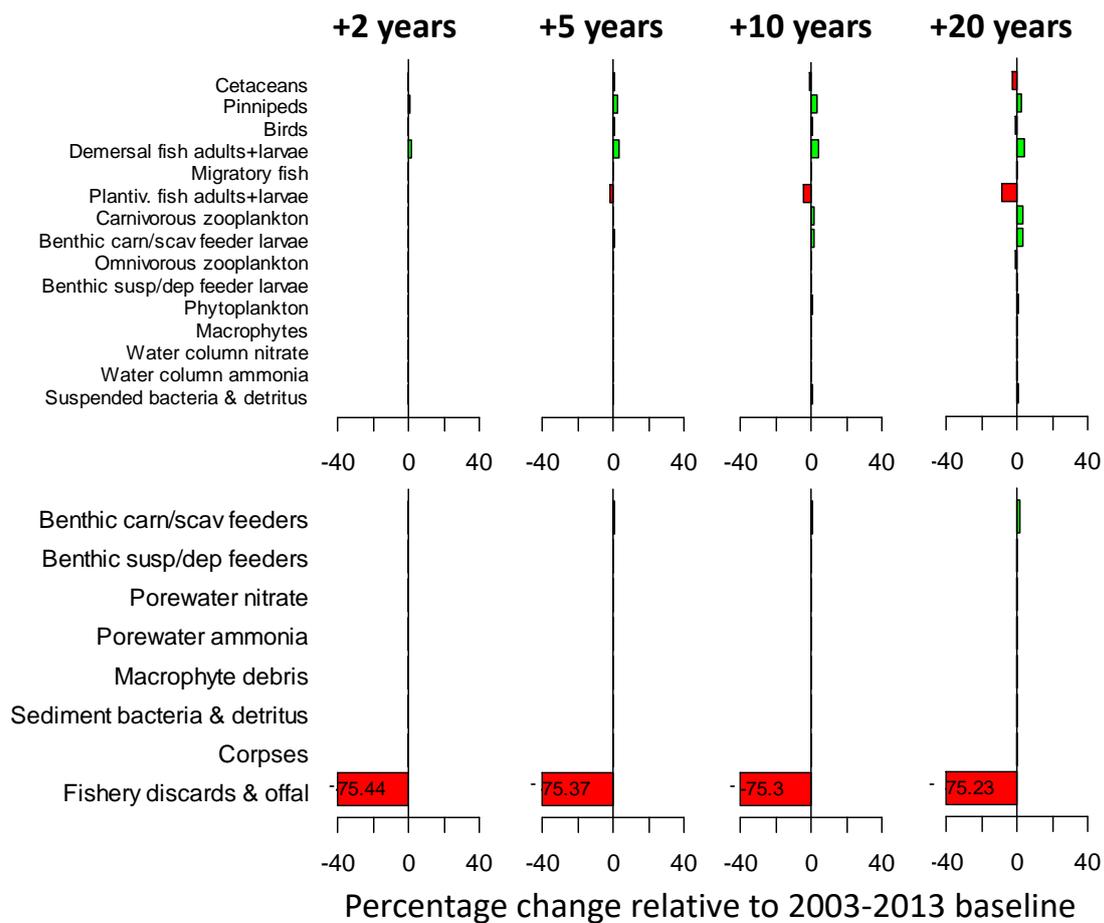


Figure 17. Simulated progression (years 2, 5, 10 and 20) of changes in the structure of the North Sea foodweb following the full implementation of the Landing Obligation ('improved selectivity') (beginning of year 1) with status-quo fishing effort (successful outcome of negotiations) relative to a baseline state in 2003-2013. Each bar represent the percentage change in annual average mass (for nutrients and detritus) and biomass for living components of the ecosystem relative to the baseline. Green bars indicate an increase, red bars a decrease relative to the baseline. Upper panel shows changes to nutrients, flora and fauna in the water column. Lower panel shows changes to nutrients and fauna in and on the seabed ('porewater' values refer to nutrient concentrations within the seabed sediments).

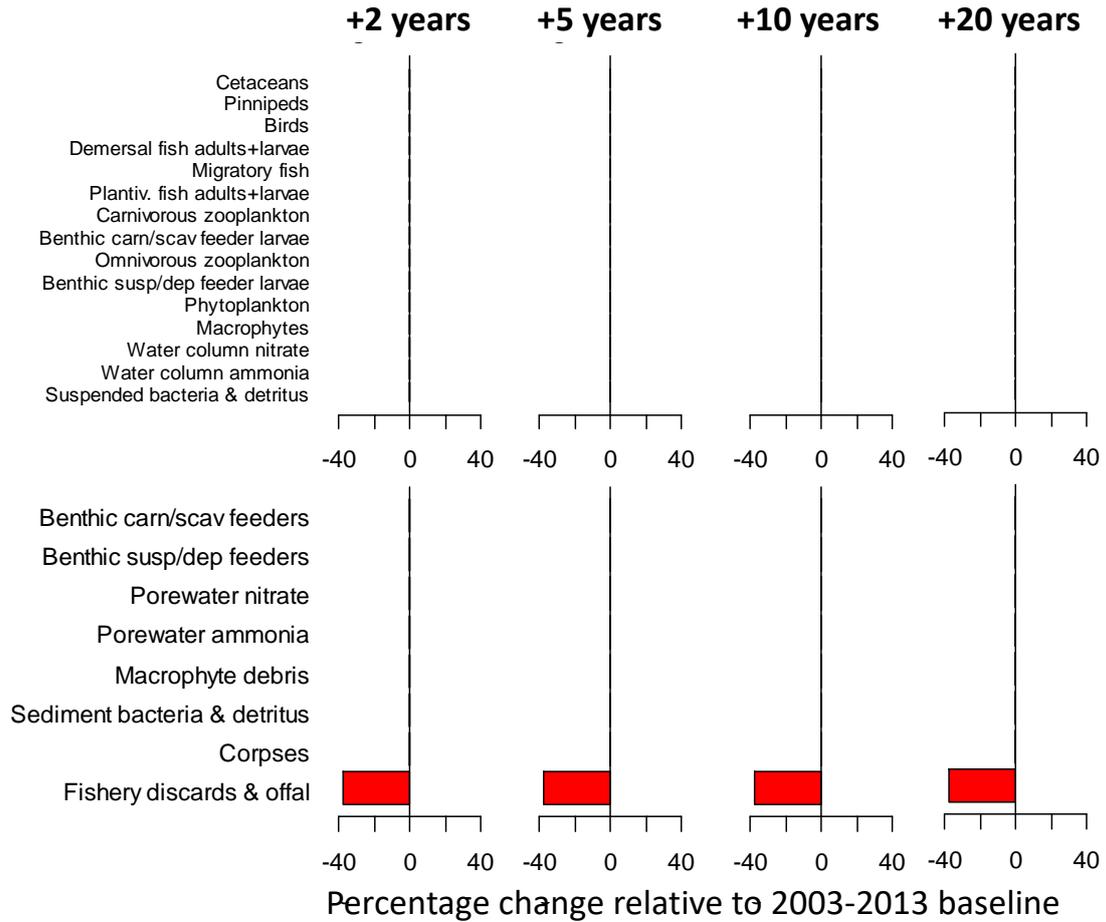


Figure 18. As Figure 17 but with the ‘only-undersize discarding’ scenario.

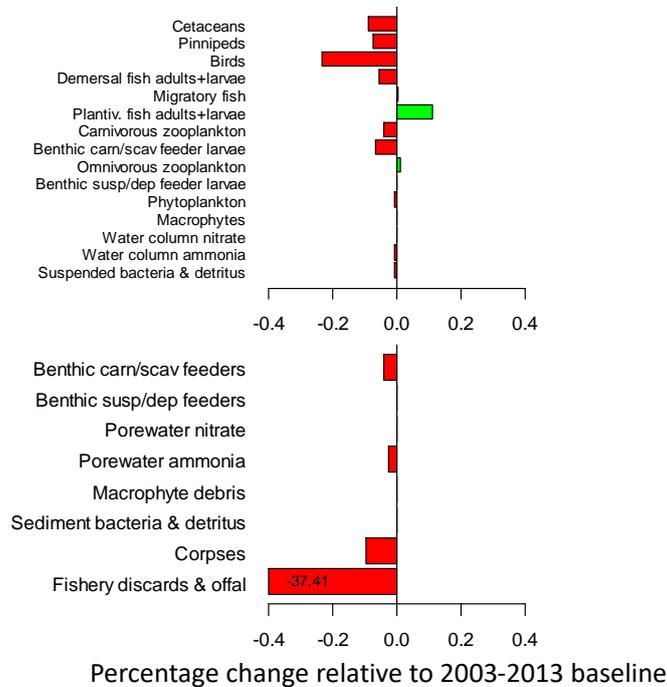


Figure 19. Magnified view of Figure 18 after 20 years. Note the 100x magnification of the percentage change scale.

Comparisons 3 and 4: effects of alternative narratives on discarding, given unilateral actions following a breakdown of Brexit negotiations.

Unsuccessful negotiations and unilateral action on TAC share by the UK amplifies the changes in the food web that are predicted to arise from the ‘improved selectivity’ discarding scenario relative to the 2003-2013 baseline (Fig. 20). Instead of maximum 10% reductions in planktivorous fish biomass after 20 years under ‘successful outcome’ activity rates, the reductions exceed 40%, with increased reductions also in cetaceans and seabirds. With ‘only undersize’ discarding, the pattern of changes in the food web are similar to those with ‘improved selectivity’ but of smaller magnitude (Fig. 21).

In terms of landings, the ‘only undersize’ discarding scenario combined with an unsuccessful negotiation outcome leads to sustained increase in demersal fish landings (Fig. 22). Initially, there is predicted to be an increase in pelagic (planktivorous) fish landings too, but over time this diminishes and eventually becomes a deficit of landings compared to the 2003-2013 baseline. On the other hand, pelagic invertebrate (squid), and crustacean (carnivore/scavenge feeding) benthos landings are expected to increase relative to the baseline state. By-catch of seabirds is predicted to increase due to the increased pelagic trawl and seine activity, whilst by-catch and whaler catch of cetaceans should decrease due to declining stock biomass. The pattern is similar but more exaggerated with the ‘improved selectivity’ discarding scenario – the increase in pelagic fish landings immediately post-negotiation breakdown is short-lived and becomes a substantial deficit by 20 years (Fig. 23).

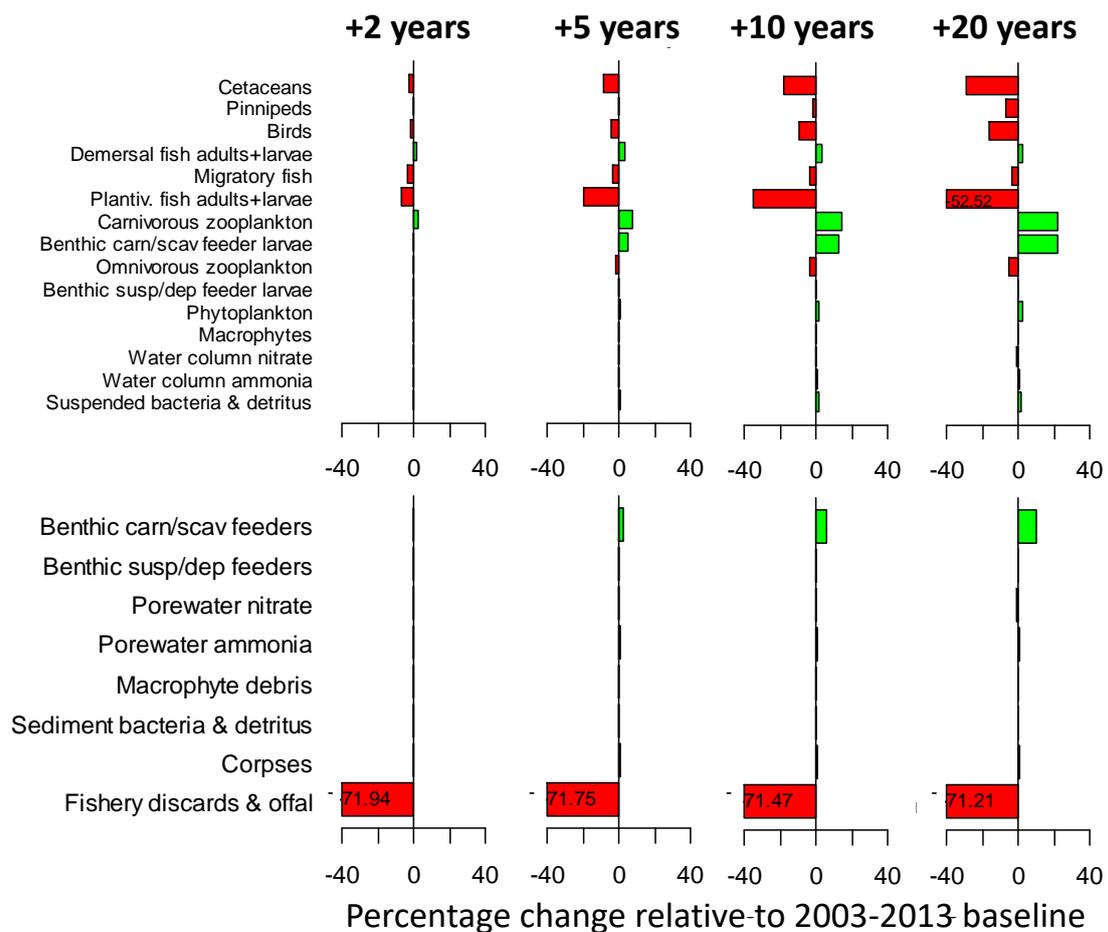


Figure 20. As Figure 17, but with the combination of full implementation of the Landing Obligation ('improved selectivity') and the post-Brexit unilateralism effort narrative.

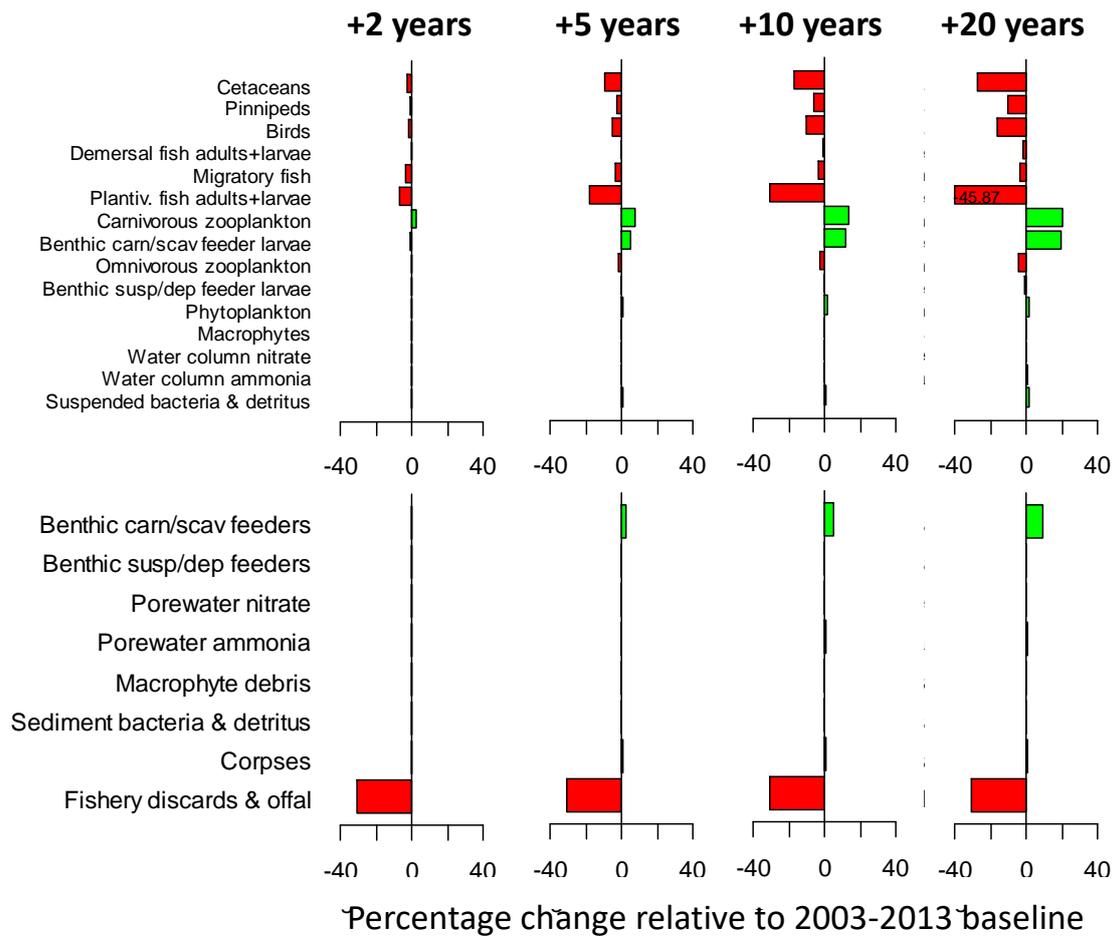


Figure 21. As Figure 18, but with the 'only-undersize' discarding scenario but the post-Brexit unilateralism effort narrative.

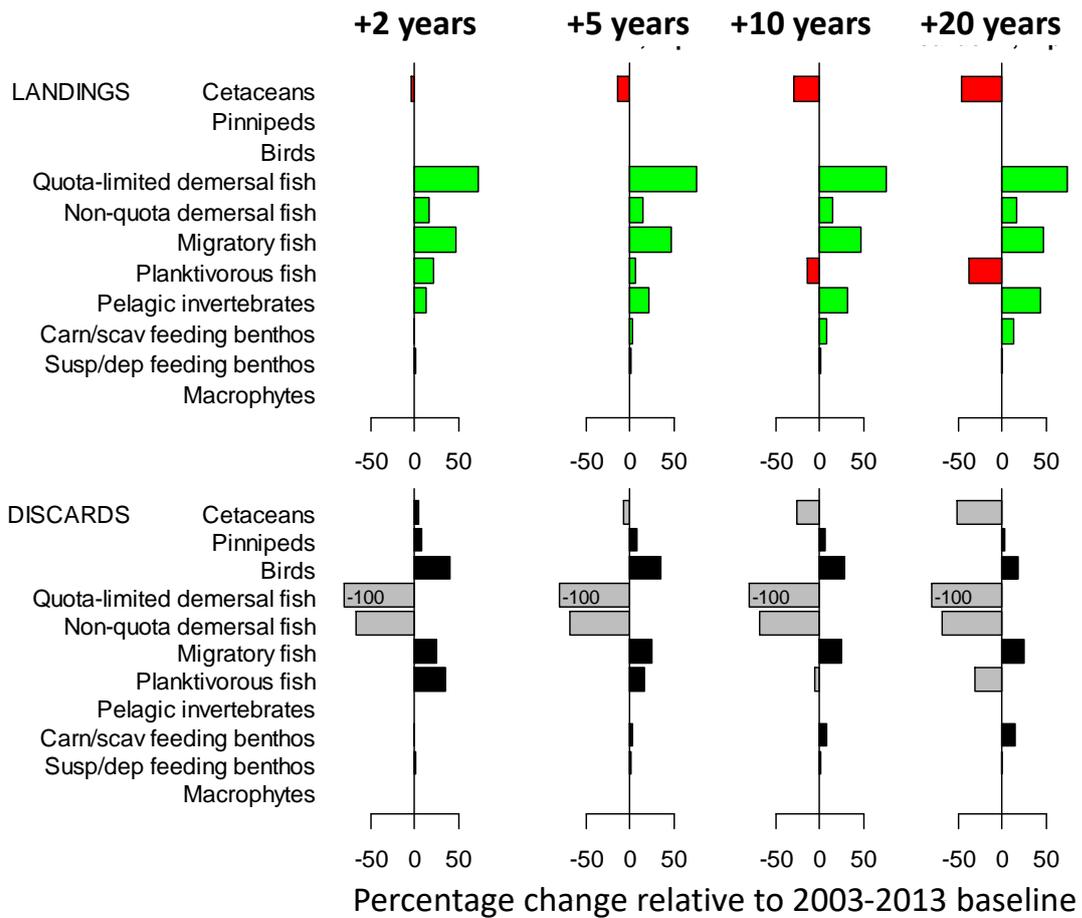


Figure 22. Simulated progression (years 2, 5, 10 and 20) of changes in the landings and discards from the North Sea food web with post-Brexit unilateralism narrative for fishing effort (unsuccessful outcome of negotiations) relative to a baseline state in 2003-2013, and full implementation of the Landing Obligation ('improved selectivity') (beginning of year 1). Each bar represents the percentage change in annual average landings (upper row, green or red bars) and discards (lower row, black or grey bars). Green bars indicate an increase in landings, red bars a decrease relative to the baseline. Black bars represent an increase in discards relative to the baseline, grey bars a decrease. For birds, seals and cetaceans, 'discards' refer to the incidental by-catch of these groups in fishing gears. The landings of cetaceans refer to catches of Minke whales taken by Norwegian whaling vessels.

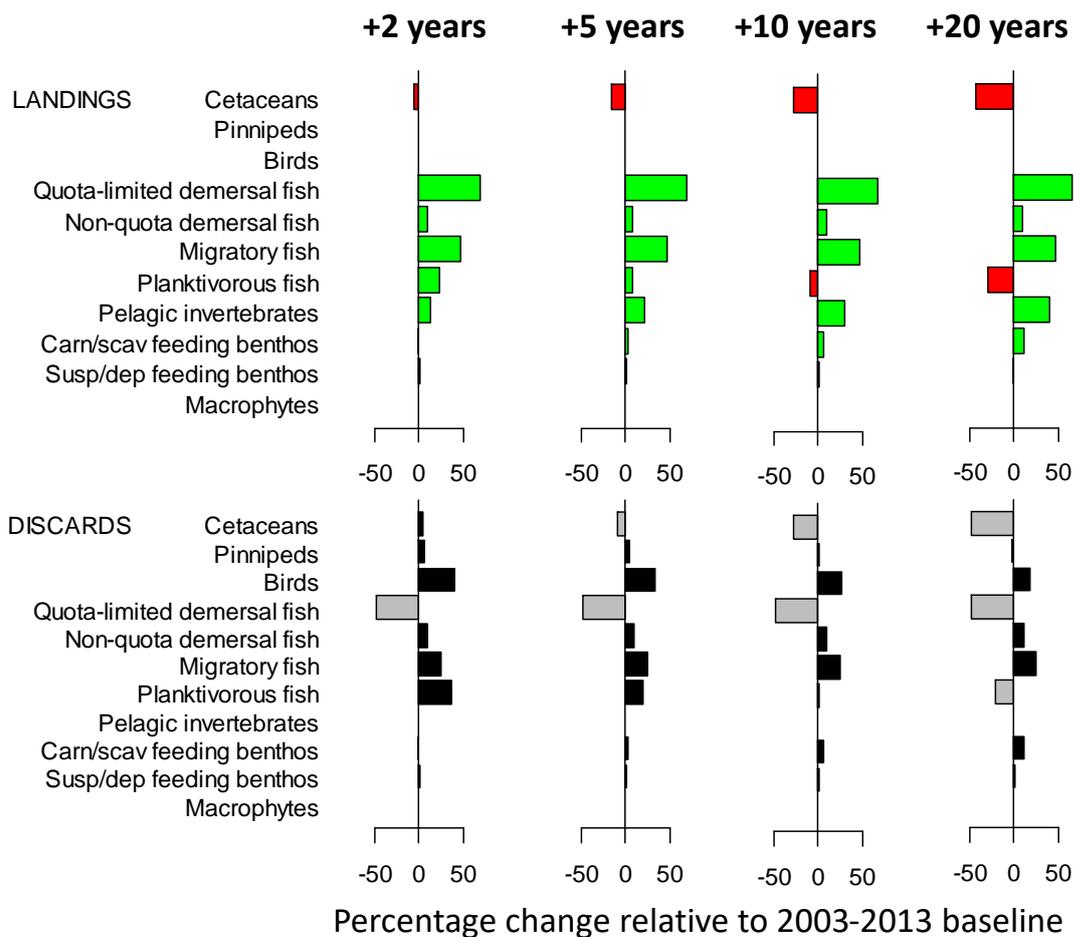


Figure 23. As Figure 22 with post-Brexit unilateralism narrative for fishing effort (unsuccessful outcome of negotiations) relative to a baseline state in 2003-2013, and the ‘only-undersize’ discarding scenario (beginning of year 1).

Comparisons 5: effects of unilateral action following breakdown of Brexit negotiations, given full implementation of the Landing obligation regardless of Brexit.

Assuming that full implementation of the Landing Obligation (improved selectivity) goes ahead regardless of Brexit negotiations, unilateral actions in the event of a breakdown were predicted to have clear negative impacts on the biomass of fish and top predators (birds, pinnipeds and cetaceans) (Fig. 24). Due to the consequent reduced predation pressure, this results in a 10-20% increase in seabed fauna (benthos groups). Initially, landings and discards of fish (other than the regulated quota-limited demersal group) and invertebrates showed an increase immediately following the onset of increases in fishing activity, but over time the depletion of planktivorous fish biomass leads to a crash (>40% reduction) in landings (Fig. 25). The reduction in predation of demersal fish larvae by planktivorous fish compensates for the increased harvesting rate, so that overall demersal fish landings are maintained at close to those in the successfully negotiated outcome.

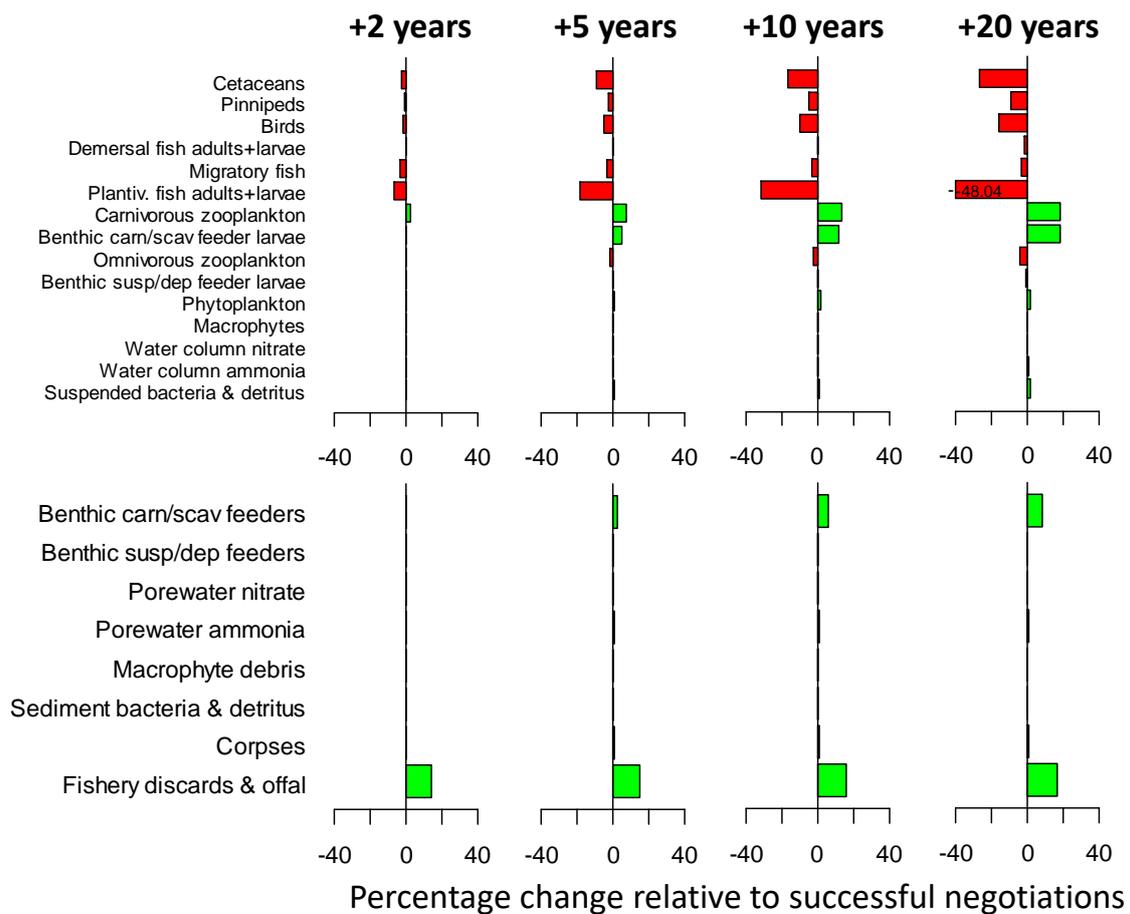


Figure 24. Simulated progression (years 2, 5, 10 and 20) of changes in the structure of the North Sea foodweb following the onset of unilateral actions due to a breakdown of Brexit negotiations(beginning of year 1), but assuming full implementation of the Landing Obligation ('improved selectivity') goes ahead regardless. Each bar represent the percentage change in annual average mass (for nutrients and detritus) and biomass for living components of the ecosystem relative to the a successfully negotiated outcome. Green bars indicate an increase, red bars a decrease. Upper panel shows changes to nutrients, flora and fauna in the water column. Lower panel shows changes to nutrients and fauna in and on the seabed ('porewater' values refer to nutrient concentrations within the seabed sediments).

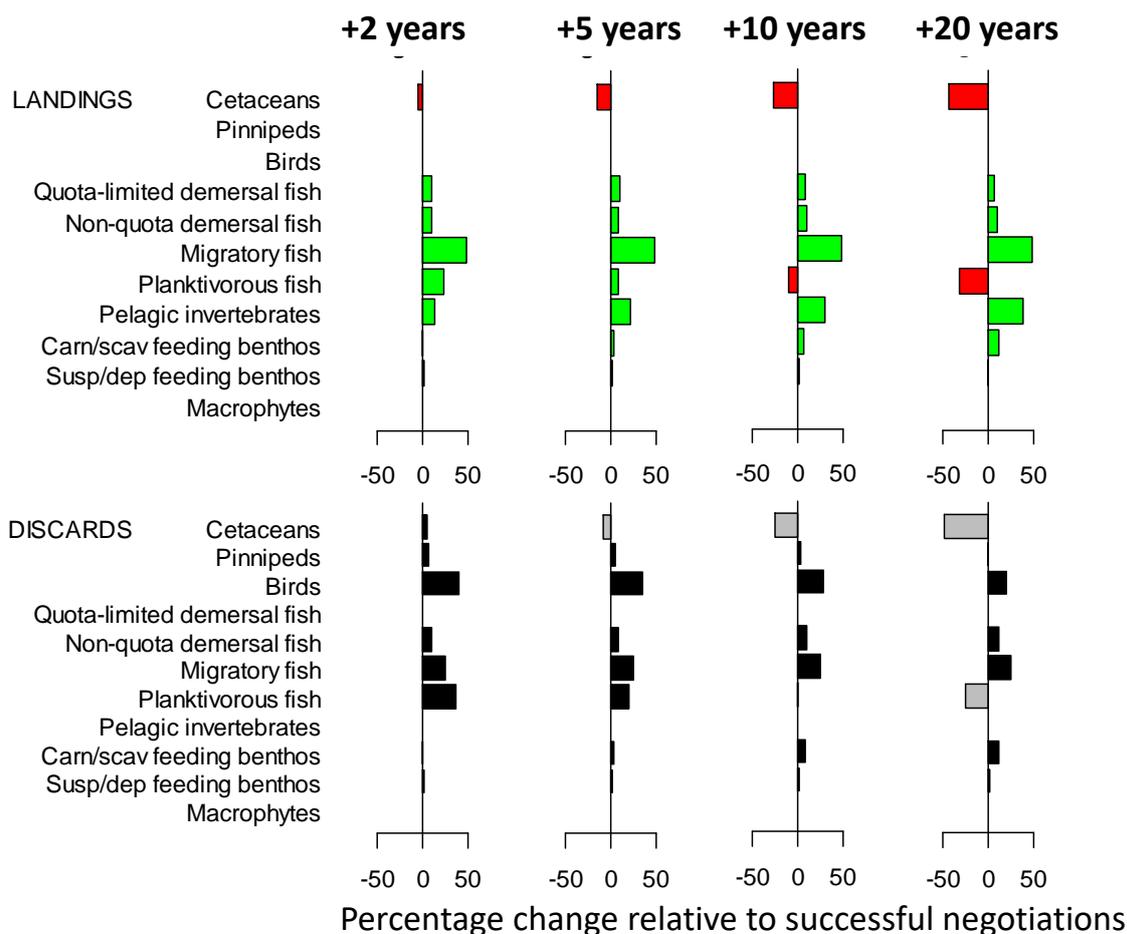


Figure 25. Simulated progression (years 2, 5, 10 and 20) of changes in the landings and discards from the North Sea following the onset of unilateral actions due to a breakdown of Brexit negotiations (beginning of year 1), but assuming full implementation of the Landing Obligation by all nations ('improved selectivity') goes ahead regardless. Each bar represents the percentage change in annual average landings (upper row, green or red bars) and discards (lower row, black or grey bars). Green bars indicate an increase in landings, red bars a decrease relative to a successfully negotiated outcome. Black bars represent an increase in discards, grey bars a decrease. For birds, seals and cetaceans, 'discards' refer to the incidental by-catch of these groups in fishing gears. The landings of cetaceans refer to catches of Minke whales taken by Norwegian whaling vessels.

Comparisons 6: effects of unilateral action following breakdown of Brexit negotiations, given partial implementation of the Landing obligation regardless of Brexit.

Assuming that only partial implementation of the Landing Obligation (continued discarding of undersize demersal fish) goes ahead regardless of Brexit negotiations, unilateral actions in the event of a breakdown were predicted to have impacts on the biomass and catches of fish and top predators (birds, pinnipeds and cetaceans) which were almost indistinguishable from those given full implementation of the Landing Obligation (Fig. 26, 27).

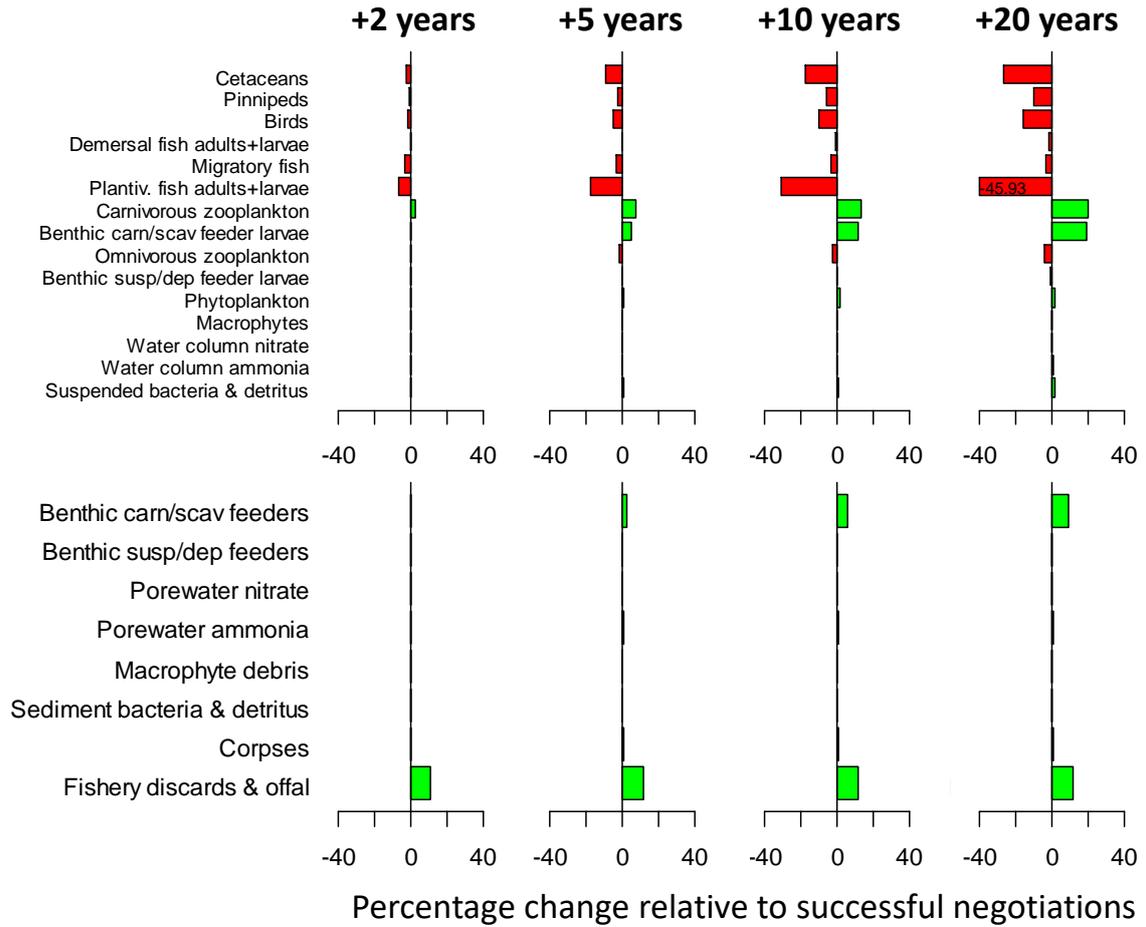


Figure 26. As Figure 24, simulated progression (years 2, 5, 10 and 20) of changes in the structure of the North Sea foodweb following the onset of unilateral actions due to a breakdown of Brexit negotiations(beginning of year 1), but assuming partial implementation of the Landing Obligation ('continued discarding of only undersize demersal fish) goes ahead regardless.

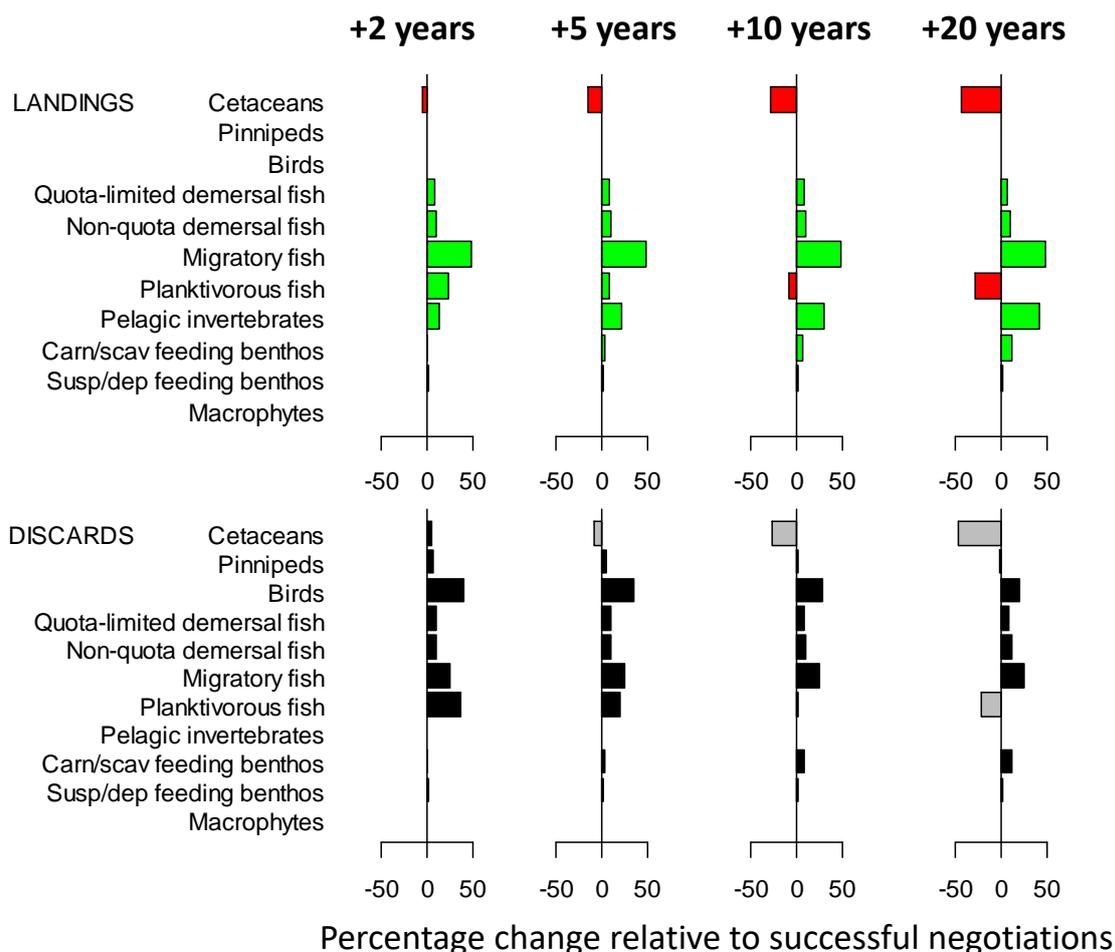


Figure 27. As Figure 25, simulated progression (years 2, 5, 10 and 20) of changes in landings and discards from the North Sea following the onset of unilateral actions due to a breakdown of Brexit negotiations(beginning of year 1), but assuming partial implementation of the Landing Obligation (continued discarding of only undersize demersal fish) goes ahead regardless.

4.2.3. Discussion

The dominant feature of all the combinations of activity and discarding scenarios is a strong interaction between pelagic (planktivorous) fish and demersal fish. Small increases in the biomass of demersal fish lead to large reductions in pelagic fish and vice-versa. This arises because as fishing mortalities have been reduced from the very high rates during the 1980s, so predation mortality by demersal fish has become a proportionately larger fraction of the overall mortality of pelagic fish. As a result, pelagic fish have become less resilient to fluctuations in other components of the ecosystem. Initial increases in pelagic fish landings immediately following the increase in fishing activity post negotiation breakdown, are short-lived and soon become a deficit compared to the 2003-2013 baseline. The decline in pelagic fish biomass under these circumstances has a universally negative effect on top-predators in the system.

Increases in demersal fish landings under the negotiation-breakdown scenarios are almost entirely due to the reductions in discarding. The premium resulting from increased demersal gear activity alone amounts to less than 10% of the baseline landings.

5. Overall Conclusions

Based on our detailed analysis of STECF data on landings, the assimilation of Norwegian landings data, Relative Stability shares, and the various existing estimates of species-specific zonal attachment, we have developed two plausible narratives for unilateralism following a hypothetical breakdown of post-Brexit negotiations on fisheries in the North Sea. We have then used these as the basis for two completely independent modelling approaches – single species stock projections, and food web modelling - to assess the risks to stock conservation and wildlife as a result of the breakdown of negotiations.

Our first narrative is based on TACs for individual stocks in the North Sea, specifically herring, cod, haddock, whiting, saithe, plaice and sole. We define plausible changes in the fishing mortality of each of these species based on the UK claiming an increased share of the TAC, over and above its existing share of landings. Note that the baseline for these changes is not Relative Stability, but the currently (pre-Brexit) realised share after taking into account existing quota swaps and trades. The rest of the EU and Norway is assumed to adhere to its existing share of the total landings. In this narrative, we are not concerned with where the catches are taken, though the underlying assumption is that non-EU vessels are excluded from the UK EEZ.

Our second narrative is not concerned with quotas and catches of individual species, but with the distribution of fishing effort by different gears. The basis for the narrative is that the UK expels all Norwegian and remaining-EU vessels from its EEZ. Norway and the EU reciprocate by expelling UK vessels. The UK then expands its effort within its EEZ to absorb that which has been expelled (except for the vacated scope for sandeel trawling), while the remaining EU and Norway vessels attempt to catch their existing quota in non-UK waters. We combine this effort re-distribution scenario with two realisations of the Landing Obligation, to which the UK officially remains committed. The first realisation is perfect implementation, and second is a probably more realistic partial implementation where undersize fish continue to be discarded but there is no over-quota discarding.

The single species stock projections, based on the first narrative, suggest that of the seven species examined, the post-Brexit scenario would affect herring and cod significantly. These two stocks have a comparatively high probability of being below the precautionary biomass reference points even at current rates of fishing. Elevated rates of fishing mortality projected under the Brexit scenario more than doubles this risk for herring and increases it by 75% for cod. Both stocks have a history of depletion and are known to be vulnerable to collapse [76,

77]. The risks are even higher in the short term, especially for cod where the current SSB is already below Bpa according to the latest ICES advice [75].

The food web modelling, based on the effort-based narrative, suggested that demersal fish in the North Sea are likely to increase over the coming 20 years, and pelagic (planktivorous) fish to decline, purely as a result of fully implementing the Landing Obligation. The Landing Obligation implies improved selectivity of demersal fishing methods so that only marketable sizes of fish are captured. As a result, the current landings of demersal fish can be achieved with lower fishing mortality rates and higher stock biomass, which should be beneficial to the fishing industry. However, increased demersal fish biomass also implies increased predation mortality on pelagic fish as an indirect effect of the Landing Obligation, so their abundances decline. This response has been exacerbated by the success of fisheries management in the past 20-30 years at reducing overall fishing mortality on demersal fish [60, 61, 62]. This means that predation mortality now represents a higher proportion of the overall mortality of pelagic fish compared to the past, and stocks become less resilient to changes in fishing mortality.

The food web model also indicates that the effect of the post-Brexit unilateralism scenario is to intensify the trends in demersal and pelagic fish abundance that result from implementation of the Landing Obligation (Fig. 20). The unilateralism narrative involves small increases in demersal fishing mortality rates which result in small changes in demersal fish landings and biomass over and above those due to the Landing Obligation. However, unilateralism also implies greater increases in pelagic fishing mortality, which these species are unable to withstand. Hence the effect of the unilateralism scenario is to amplify the indirect impact of the Landing Obligation on pelagic fish.

Indirect consequences for non-target groups of species as a result of the changes in fishing under combinations of Landing Obligation and post-Brexit unilateralism, are reduced biomasses of cetaceans and seabirds, and increased biomasses of demersal fish, pelagic invertebrates (krill, squid and other macro-plankton), and of carnivorous and scavenge feeding benthos (mainly crustaceans such as shrimps, prawns and crabs). Declines in both cetaceans and seabirds in the model arise from the combination of decreasing food supply (planktivorous fish) and increased mortality due to by-catch in fishing gears [82]. Seals are predicted to be less vulnerable to the projected decline in planktivorous fish due to their greater feeding preference for demersal fish.

Our findings are contingent on the assumption that the UK does not attempt to establish a sandeel fishery in its EEZ following exclusion of foreign vessels from its waters. Currently, Norway and Denmark catch over 150,000 tonnes of sandeels per year (2003-2013 average) in the UK North Sea EEZ while the UK catches only 2000 tonnes. It seems unrealistic to suppose that the UK could develop the means to catch and process this resource in the short term, even if markets were available and it was supported by policy (but see [83]). Large areas of the UK EEZ are explicitly closed to sandeel fishing to protect wildlife, in particular breeding seabirds [84], and this will most likely remain, or even be expanded as a policy

commitment. However, if we were to factor the establishment of a post-Brexit UK sandeel fishery into our models, in addition to the existing quota caught by Norway and Denmark continuing to be taken but outside UK waters, then the consequences for the ecosystem would clearly be significantly exacerbated and we could at least expect more severe impacts on pelagic (planktivorous) fish and indirect effects on cetaceans and birds. On the other hand, it might be asserted that under UNCLOS the UK is required to offer the EU and Norway the opportunity to exploit shared stocks of sandeels in its waters if it is unable to do so itself for whatever reason. So, there is considerable uncertainty over the outcome of either negotiated or unilateral outcomes for sandeels.

In conclusion, both of our modelling approaches are convergent in that they identify the greatest threat from unilateralism and a breakdown of fisheries negotiations is to stocks of pelagic (planktivorous) fish, especially herring. Risks to demersal stocks as a whole are smaller, though the risk to cod is significantly increased. In terms of consequences for wildlife, the main risks are to cetaceans and seabirds.

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APPENDIX 1. Landings and effort data and analysis methods

Processing of FAO/ICES official landings statistics.

Official data on annual landings of fish are available from ICES (<http://www.ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx> ; note that although the data are referred to as catch statistics, they in fact only reflect landings as discards are not included). The background to the data have been documented in an ICES Cooperative Research Report [1.1]. Briefly, the data are available in three tranches: 1903-1948; 1950-2010; 2006-2016. However, constructing a coherent time-series through the entire period 1903-2016 is notoriously difficult due to a variety of factors:

- The early data up to 1950 may be partial, having been recovered from print-published reports (Bulletin Statistique).
- Until 2010, the conventions for recording capture location and species identities have been variable between countries, and even within countries over time. For example, there may be several synonyms in the data for any one species. Also, the spatial resolution of the data is organised around the FAO/ICES statistical areas – for example, ICES area VII (now referred to as FAO area 27.7; Celtic Sea) is divided into 10 divisions, some with further subdivisions. However, the data may be assigned to unique divisions, subdivisions, or combinations of these, or just to area VII as a whole. This means that it is practically impossible to generate meaningful syntheses of the data at the finest spatial resolution.
- Since 2010, a greater degree of rigor has been enforced on the data records, so that synonyms are reduced (but not eliminated) and the spatial structure is more organised, with well documented levels of aggregation within the dataset.

Over a period of years, we have developed a database linking synonyms for species across the three tranches of data, and linkages between species identities and coarser ecologically significant groups [1.2]. We have also identified a minimum practical spatial resolution for aggregating across the spatial identities of the landings records while preserving the integrity of the data. This has largely automated the process of merging data across the three tranches.

For this project, we aggregated the annual landed weights of minimally resolvable species identities across 5 spatial areas covering the continental shelf around the UK (Table A1.1). We did not aim to cover the off-shelf waters west of the UK which may nevertheless lie within the UK EEZ (e.g. around Rockall).

Table A1.1. Spatial aggregation rules for processing and merging the three tranches of FAO/OCES landings data spanning 1903-2016. The rules are not perfect (for example VII d-k is allocated to the English Channel), but such infelicities are few and minor. There are no spatial designations relevant to these coarse regions which are unallocated in the database.

| Aggregated area | ICES area designations and synonyms included (1903 – 2005) | FAO area designations included (2006-) |
|------------------|--|---|
| North Sea | IV IVa IVa,b IVc IV a IV b IV c IV (not specified) IV a+b (not specified) IIIa and IVa+b (not specified) IIIa and IV (not specified) IV b+c (not specified) IIIa,IV | 27.4.b 27.4.a 27.4.c |
| West of Scotland | VI (not specified) VI VI a VIa Via | 27.6.a |
| Irish Sea | VII a VIIa VII a-f (not specified) VIIa,f | 27.7.a |
| Celtic Sea | VII VII (not specified) VII b VIIb-c VIIb,c VIIb,c +g-k VII f VIIf VII f-k (not specified) VII g VIIg-k VII g-k (not specified) VII h VII j (not specified) VII j2 | 27.7.b 27.7.f 27.7.g 27.7.h 27.7.j.1 27.7.j.2 27.7.j_NK |

| | | |
|-----------------|--|------------------|
| | VII j1 | |
| English Channel | VII d VII d+e (not specified) VII d-k (not specified) VII e VII d-e VII d,e | 27.7.d 27.7.e |

The FAO species reference list includes 11,564 species identities (including synonyms). The three tranches of data spanning 1903-2016 include 838 identities including spelling variants and synonyms. From these we can identify 594 unique identities, of which 30 make up around 95% of the weight of fish landed from the areas around the UK.

We identified 6 coarse groupings of species based on feeding and migration patterns (Table A1.2), and developed the linkages from every species identity in the dataset based on ecological information in the literature and in the FishBase website (www.fishbase.org)

Table A1.2. Groupings of species in the FAO/ICES landings data into coarse functional groups

| Group name | Description and archetypes | Number of unique entities in the FAO/ICES dataset |
|--------------------------------|---|---|
| Planktivorous | Predominantly plankton-feeding mid-water or near-seabed fish. May be migratory but within the confines of typical ICES areas: Herring, sprat, sandeel, sardine, Norway pout | 84 |
| Migratory pelagic | Mainly plankton but also small fish-eating species which undertake large scale migrations spanning multiple ICES areas: Mackerel, Horse mackerel, Blue whiting, tunas | 20 |
| Demersal | Mostly bottom-living, mostly fish and seabed fauna (benthos) eating fish which are largely resident within an ICES area: Cod, haddock, whiting, plaice, sole, saithe, anglerfish | 295 |
| Benthos filter/deposit feeders | Seabed living invertebrates which feed mainly by sieving plankton and detritus out of the water or by processing seabed sediments. Mainly molluscs: Scallop, mussels, oysters, clams | 81 |
| Benthos carnivore/scavenge | Seabed living invertebrates which are predatory on other animals (or even fish), and/or carrion-eaters | 62 |

| | | |
|-----------------------|---|----|
| feeders | consuming corpses and discards. Mainly crustaceans: Nephrops, crabs, lobsters, prawns, shrimps | |
| Pelagic invertebrates | Pelagic invertebrates: Squid, cuttlefish | 29 |

Scientific, Technical and Economic Committee for Fisheries (STECF) and Norwegian landings data analysis

STECF is part of the EU Joint Research Centre, and coordinates the gathering of all aspects of data in support of the Common Fisheries Policy. STECF make publically available all data from on the landings, discards, effort and economic performance of the fleet sectors of all EU countries (<https://stecf.jrc.ec.europa.eu/dd/effort>). From 2000 onwards, the landings and effort data are resolved by at least 1 longitude x ½ latitude cells (approximately 30 x 30 nautical miles). Discard data are available only at a more aggregated spatial resolution.

The STECF data are highly amenable to analysis of the distributions of landings and effort by different fleet sectors in relation to any user-defined geographic boundaries. Several recent projects related to understanding the implications of Brexit for fisheries have exploited these data (e.g. [1.3, 1.4]).

The STECF dataset for NW European water includes records on 101 different species covering mostly finfish. Invertebrates are under-represented in the records. We used the same aggregation rules as for the FAO landings data to integrate these species into the 6 coarse ‘functional’ categories where appropriate.

The data contain 32 different fishing gear designations. Some of which are local variants appropriate to particular countries or regions. We aggregated the STECF gear types up into 11 coarser groups for use in our further modelling of the implications of enforced changes in fishing patters. The aggregation rules are shown in Table A1.3

Table A1.3 Correspondence between raw STECF gear codes and gear categories in the analyses presented here.

| STECF Code | Gear description | StrathE2E model gear type |
|----------------|---|---------------------------|
| PELAGIC TRAWLS | Pelagic trawls | Pelagic trawls & seines |
| PEL_TRAWL | Pelagic Trawl | |
| PEL_SEINE | Pelagic seine nets | |
| TR3 | Bottom trawls and seines of mesh size equal to or larger than 16 mm and less than 32 mm – mostly targeting sprat. | Sandeel & sprat trawls |
| OTTER | Bottom trawls (for sandeel) | |

| | | |
|---|---|-----------------------------------|
| LL (landings in a statistical rectangle >50% by weight mackerel) | Drifting longlines (for pelagic fish) | Longline mackerel |
| BT2 | Beam trawls of mesh equal to or larger than 80 mm and less than 120 mm. | Beam trawl demersal |
| BT1 | Beam trawls of mesh equal to or larger than 120 mm | |
| DEM_SEINE | Danish and Scottish seiners | Demersal seine |
| TR1 | demersal trawls/seines with larger mesh sizes > 100MM | Demersal otter trawl (mainly TR1) |
| TR2 (landings in a statistical rectangle <30% by weight Norway lobster) | Demersal trawls and seines with mesh 70-99mm | |
| BOTTOM TRAWLS | Bottom trawls | |
| 3A | Bottom trawler mesh size \geq 32 mm) | Longline & gillnets demersal |
| LL (landings in a statistical rectangle <50% by weight mackerel) | Set longlines (for demersal fish) | |
| GN1 | Gill nets, entangling nets. | |
| GILL | Drift and fixed Nets except Trammel Nets | |
| 3B | Gillnet \geq 60 mm | |
| TRAMMEL | Trammel nets | |
| GT1 | Trammel nets | |
| BEAM | Beam trawl targeting shrimp (in the North Sea) | |
| TR2 (landings in a statistical rectangle >30% by weight Norway lobster) | Demersal trawls and seines with mesh 70-99mm | Nephrops trawl |
| POTS | Pots and traps | Creels and pots |
| DREDGE | Dredges (targeting scallops in the North Sea) | Mollusc dredges |

There were some specific considerations involved in the gear aggregations, depending on the species targeted in specific areas.

Beam trawls: The STECF beam trawl categories BT1 and BT2 more or less exclusively target demersal finfish species in the North Sea. It is therefore reasonable to combine BT1 and BT2 into a single model gear. However, care must be taken with the BEAM class. In the North Sea, landings from BEAM gears are more or less exclusively common shrimp (98.7% of the total). Elsewhere, landings from BEAM are more or less exclusively demersal fish. We therefore created a ‘Beam trawl shrimp’ gear in the North Sea.

TR2 trawls: The TR2 category covers gears used in a variety of fisheries in the North Sea:

- A fishery for Nephrops, which has a significant bycatch of demersal fish
- Mixed fishery in the southern North Sea, with whiting and other finfish species as the main components
- Danish and Swedish fishery targeting demersal finfish in the Skagerrak.

Since the targeted Nephrops fishery operates exclusively in muddy areas and there are particular concerns about the seabed impact of this fishery we sought to disaggregate TR2 to identify the Nephrops trawl component. Country level landings data help us with the disaggregation. If the TR2 landings by an individual country from an individual ICES statistical rectangle comprised more than 30% Nephrops then we assigned that rectangle’s TR2 landings and activity to the Nephrops gear. If it was less than 30% we assigned it to the demersal otter trawl category.

Longlines (LL): Longlines are extensively used in drifting, near-surface set mode for catching mainly mackerel and tuna, and in a near-seabed set mode for catching e.g. cod and ling. Apart from the very different species targeting of these two modes of operation, there are consequences for by-catch of non-target species. In particular, various seabird species are vulnerable to

A major shortcoming of using the STECF data in the context of Brexit-related analyses of the distribution of fishing effort and landings is that it includes data only from EU Member States. So, while effort and landings from EU nation activity in Norwegian and Faroese territorial waters are included, the equivalent data for Norwegian and Faeroese vessels are not. This is a major problem for analysis of spatial and national shares of total yields and effort in the North Sea where Norway has a very significant share of the total catch and a significant portion of the EEZ territory. The Faeroe Islands also have an access agreement with the EU, but their activities in the North Sea are relatively minor. Faeroese vessels are, however, significant in relation to catches of mackerel and Blue whiting off the west of Scotland.

Both Needle (2015) [1.4] and Napier (2018) [1.3] recognise the lack of Norwegian data in their analyses. Similarly, the analyses of zonal attachment based on landings in the Defra

White Paper [1.5] have data-voids corresponding to Norwegian water. While part of the CFP, the UK has had no requirement to gather of its own on fish that are extracted from its EEZ by non-EU nations but not landed at UK ports. So, we made a request to the Norwegian Directorate of Fisheries, Statistics Department, who kindly provided all the Norwegian annual landings data from the North Sea and west of Scotland regions, resolved by species and 1 longitude x ½ latitude cells for the years 2003-2016.

We processed the Norwegian data to conform with the STECF data, but still we lacked a breakdown of the landings by gear, or any record of the effort expended by Norwegian vessels to compare with the EU effort data. However, given that each of the STECF gears largely targets particular species (OTTER targets sandeels, TR3 targets sprat, etc), and assuming that the pattern of targeting and the selectivity of the gear types is the same in the EU and Norwegian fishery, we developed a scheme to impute the Norwegian effort, and the distribution of Norwegian landings across gear types for a given year and geographic area (Figure A1.1).

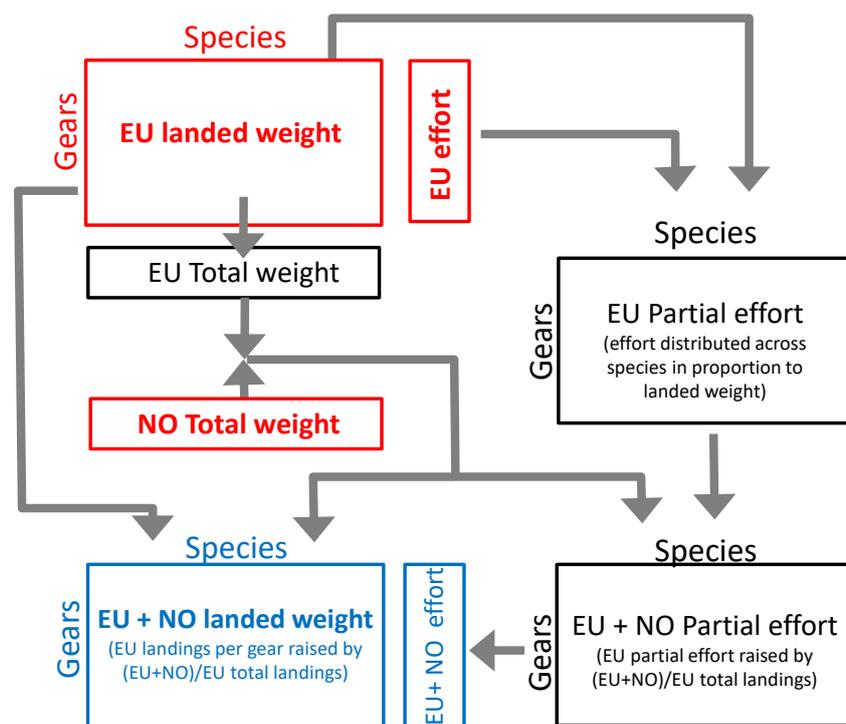


Figure A1.1. Workflow for imputing Norwegian effort per gear type, and the distribution of Norwegian landings across gear types in a given year and geographic area, given the STECF data and the Norwegian landings data obtained from the Directorate of Fisheries. Red cells indicate the data that we have from STECF and the Norwegian Fisheries Directorate, blue cells indicate the data we wish to impute. EU = European Union fleet data, NO = Norwegian fleet data. Estimates of Norwegian effort and landings per gear alone, is simply the imputed total (EU+NO) minus the known EU component.

References for Appendix 1

[1.1] Lassen, H., Cross, D., and Christiansen, E. 2012. One Hundred Years of Catch Statistics for the Northeast Atlantic. *ICES Cooperative Research Report* **311**. 21 pp.

[1.2] Heath, M.R. 2005. Regional variability in the trophic requirements of shelf sea fisheries in the Northeast Atlantic, 1973-2000. *ICES Journal of Marine Science* **62**, 1233-1244.

[1.3] Napier, I.R. 2018. The potential value to the UK fishing fleet of larger shares of the landings from the UK EEZ. NAFC Marine Centre, University of the Highlands and Islands, Port Arthur, 2018. www.nafc.uhi.ac.uk/eez-reports

[1.4] Needle, C. 2015. FIS005 - Reconsideration of European Relative Stability quota shares and implications for the Landings Obligation. A study commissioned by Fisheries Innovation Scotland (FIS) <http://www.fiscot.org/> 30pp.

[1.5] Defra (Department of Environment Food and Rural Affairs) 2018. Fisheries white paper: Sustainable fisheries for future generations. 60pp.
https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/722074/fisheries-wp-consult-document.pdf

APPENDIX 2. Additional Figures on landings history for Section 2

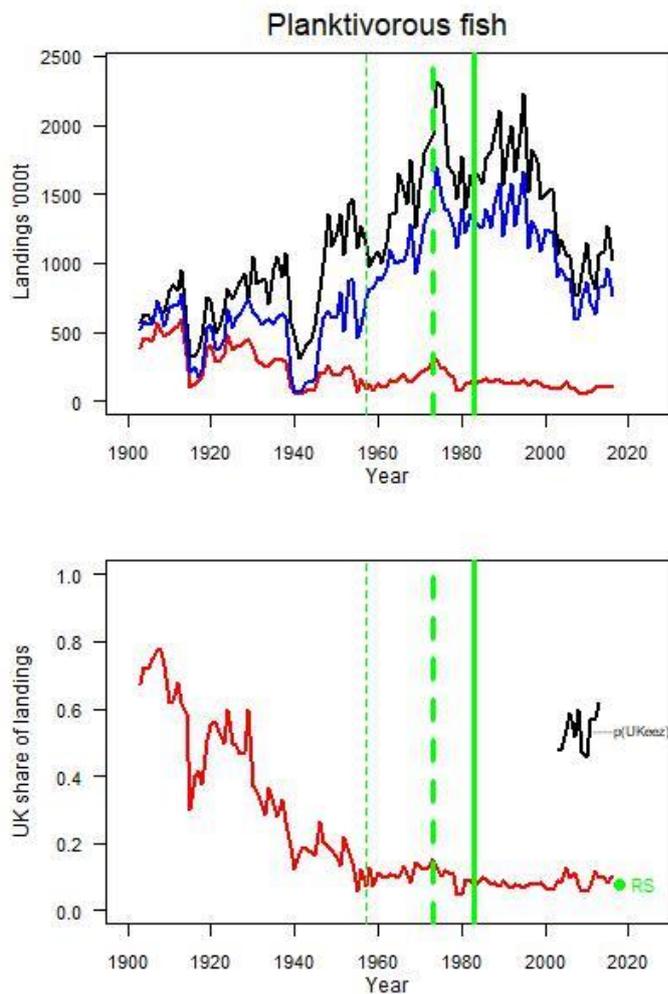


Figure A2.1. Landings history for all regions around the UK and all planktivorous fish species combined (e.g. herring, sprat, sandeels). Top panel: Black line shows the total international annual landings; blue line shows the landings by the combination of all nations currently members of the EU28 (including the UK); red line shows the landings by the UK. The solid vertical green line indicates the start of the CFP; heavy vertical dashed green line shows when the UK joined the EEC, and the light vertical green dashed line shows the formation of the EEC by the Treaty of Rome. Lower panel shows the proportion of total international landings due to the UK (red line). The green symbol labelled RS indicates the UK's Relative Stability share of all EU managed planktivorous fish species in 2018. The short black line spanning 2003–2013, labelled 'p(UKeez)' indicates the proportion of total international landings taken in the UK exclusive economic zone.

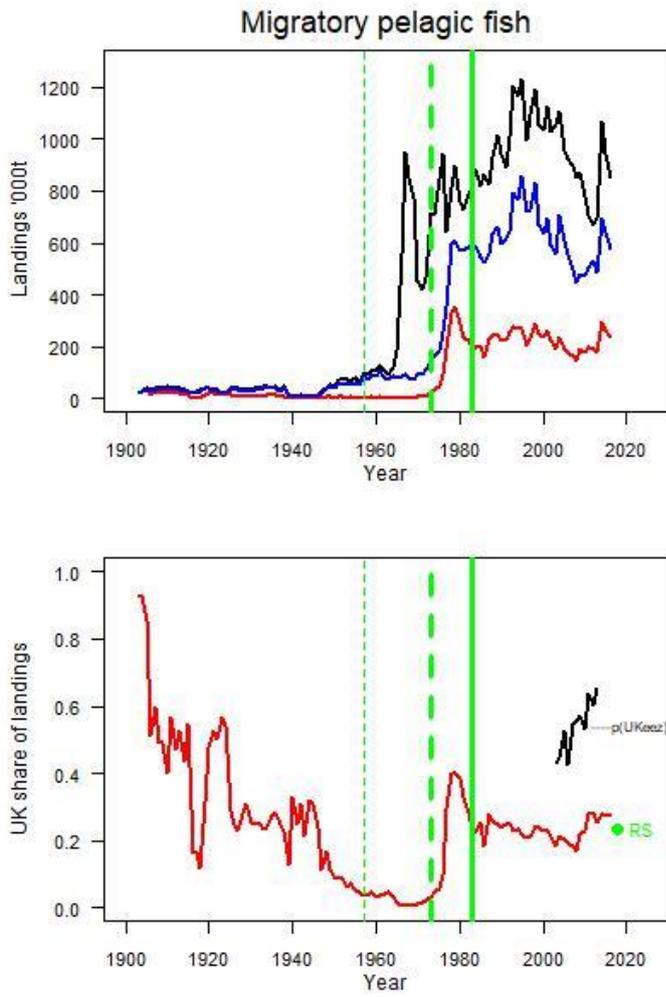


Figure A2.2. Landings history for all regions around the UK and all migratory fish species combined (e.g. blue whiting, mackerel, horse mackerel). Details as in Figure A2.1.

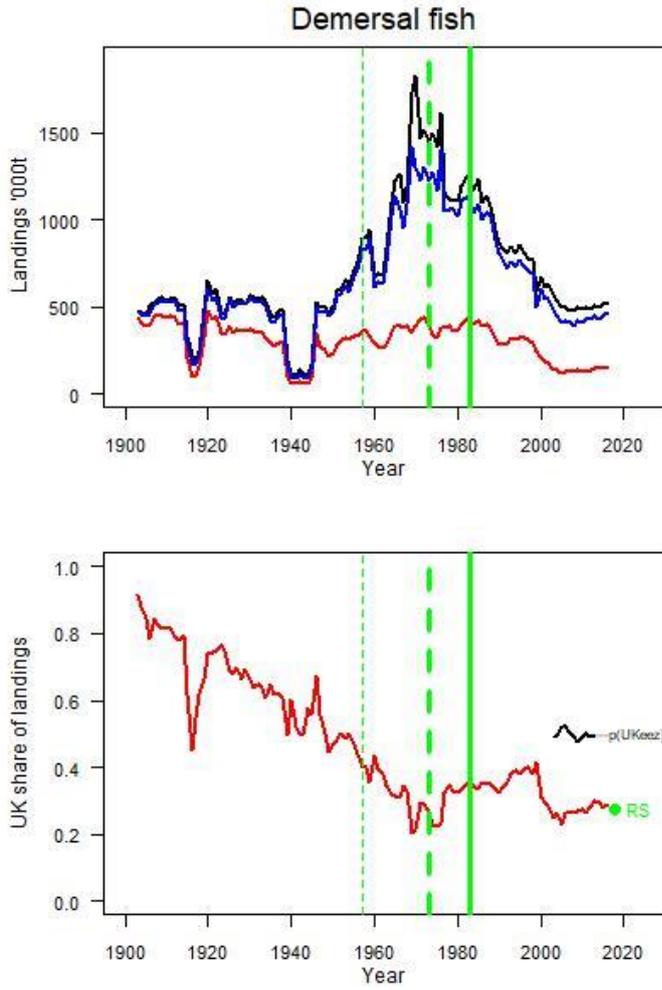


Figure A2.3. Landings history for all regions around the UK and all demersal fish species combined (e.g. cod, haddock, whiting, plaice, sole). Details as in Figure A2.1.

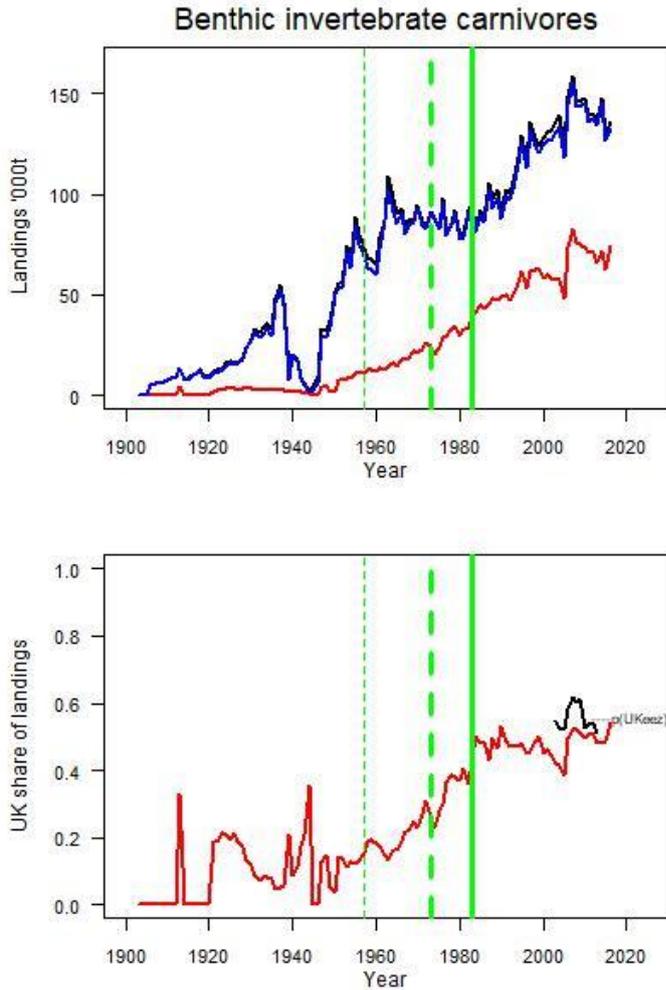


Figure A2.4. Landings history for all regions around the UK and all carnivorous feeding shellfish species combined (e.g. crustaceans such as crabs, lobsters, prawns and Norway lobster (*Nephrops*)). Details as in Figure A2.1, except that in this case we cannot estimate the Relative Stability fraction of the whole group that has been landed from UK water since of all the species involved only Norway lobster and Northern prawn are subject to EU CFP TACs and quotas. The data on proportion of landings from UK waters shown here (based in the STECF database, similarly represent only these species so are only a very partial reflection of the true proportions of the group as a whole taken from UK waters).

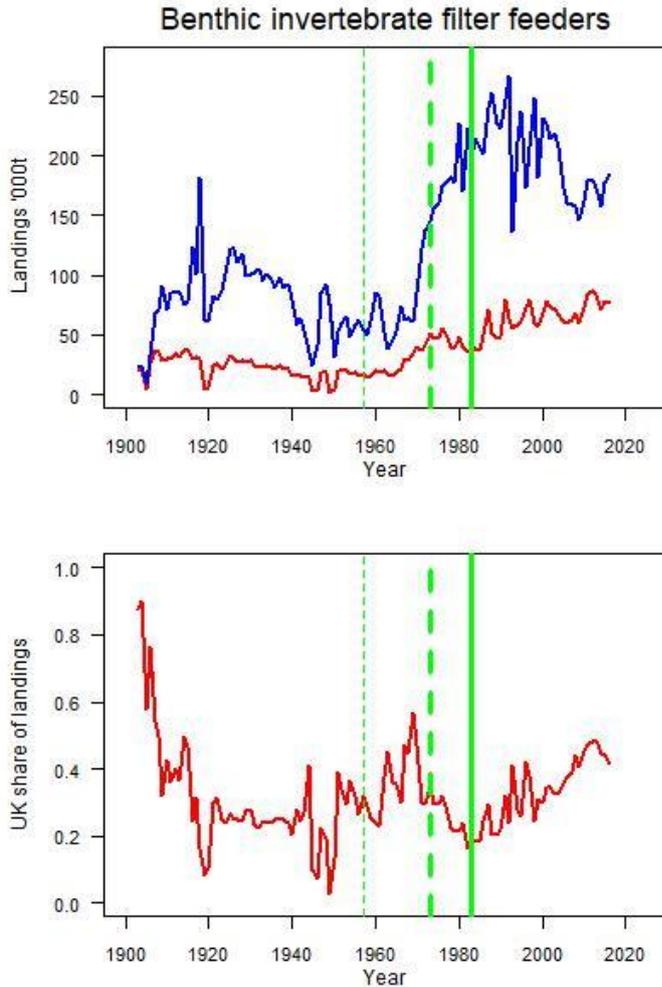


Figure A2.5. Landings history for all regions around the UK and all filter feeding shellfish species combined (e.g. molluscs such as scallops, mussels, oysters, clams). Details as in Figure A2.1, except that in this case we cannot estimate a Relative Stability fraction of the whole group since none are subject to EU CFP TACs and quotas. Similarly, of all species involved, only the Great Atlantic scallop is included in the STECF database so we cannot present a meaningful estimate of the proportion taken from UK waters.

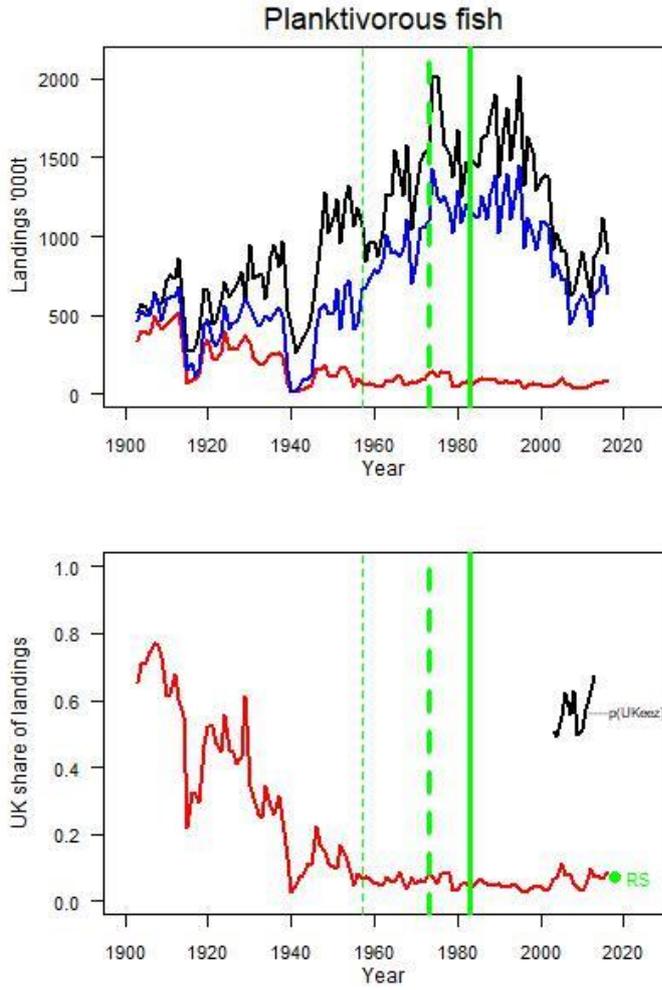


Figure A2.6. Landings history for planktivorous fish species (e.g. herring, sprat, sandeels) in the North Sea (FAO area 27.4). Details as in Figure A2.1.

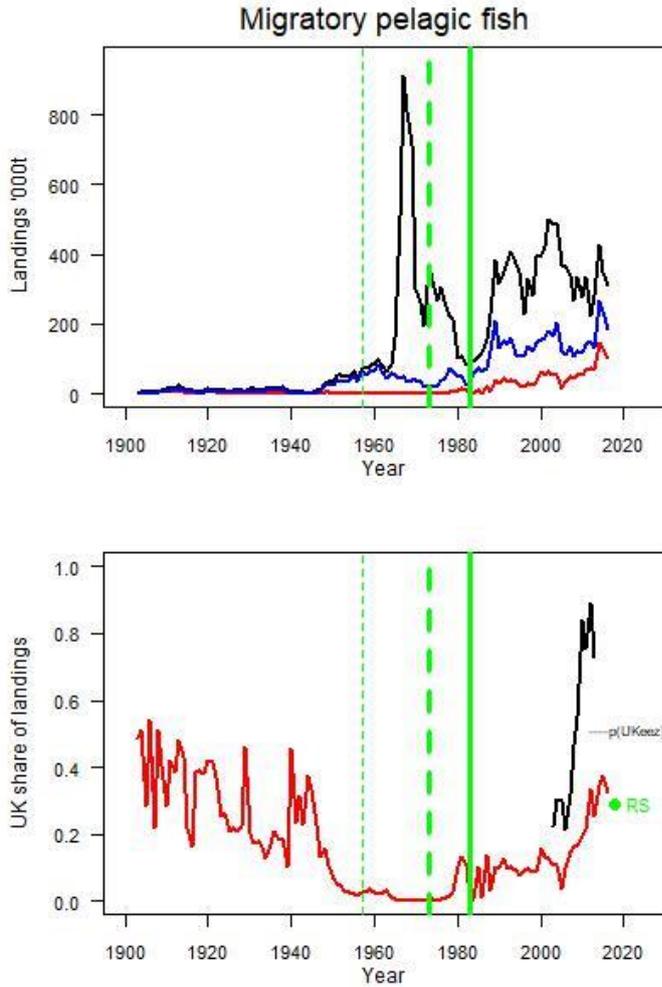


Figure A2.7. Landings history for migratory fish species (e.g. blue whiting, mackerel, horse mackerel) in the North Sea (FAO area 27.4). Details as in Figure A2.1.

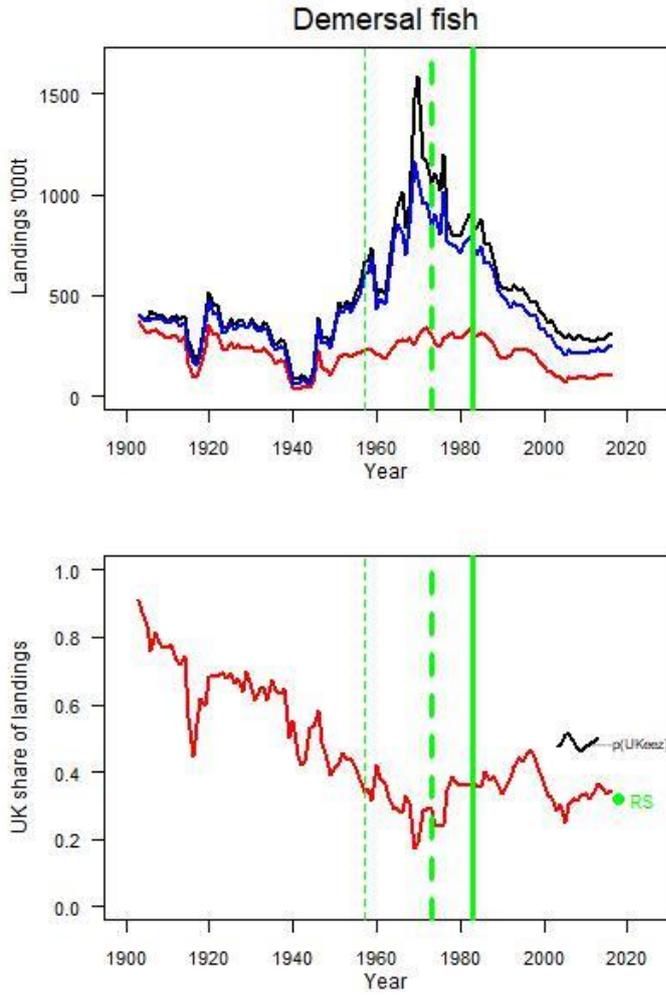


Figure A2.8. Landings history for demersal fish species (e.g. cod, haddock, whiting, plaice, sole) in the North Sea (FAO area 27.4). Details as in Figure A2.1.

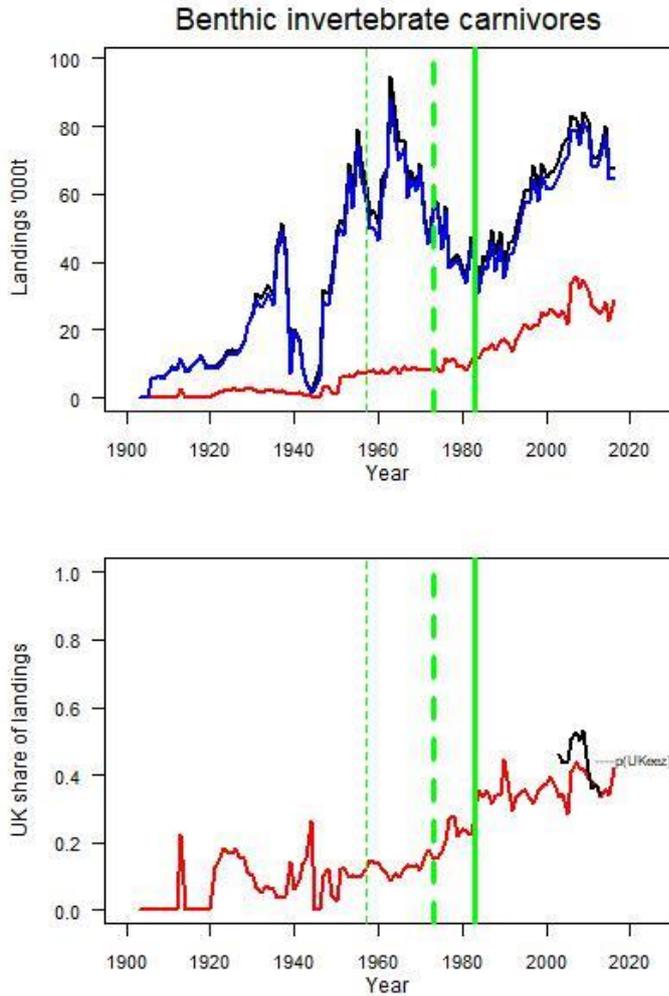


Figure A2.9. Landings history for all carnivorous feeding shellfish species combined (e.g. crustaceans such as crabs, lobsters, prawns and Norway lobster (Nephrops)) in the North Sea (FAO area 27.4). Details as in Figure A2.1, except that in this case we cannot estimate the Relative Stability fraction of the whole group that has been landed from UK water since of all the species involved only Norway lobster and Northern prawn are subject to EU CFP TACs and quotas. The data on proportion of landings from UK waters shown here (based in the STECF database, similarly represent only these species so are only a very partial reflection of the true proportions of the group as a whole taken from UK waters.

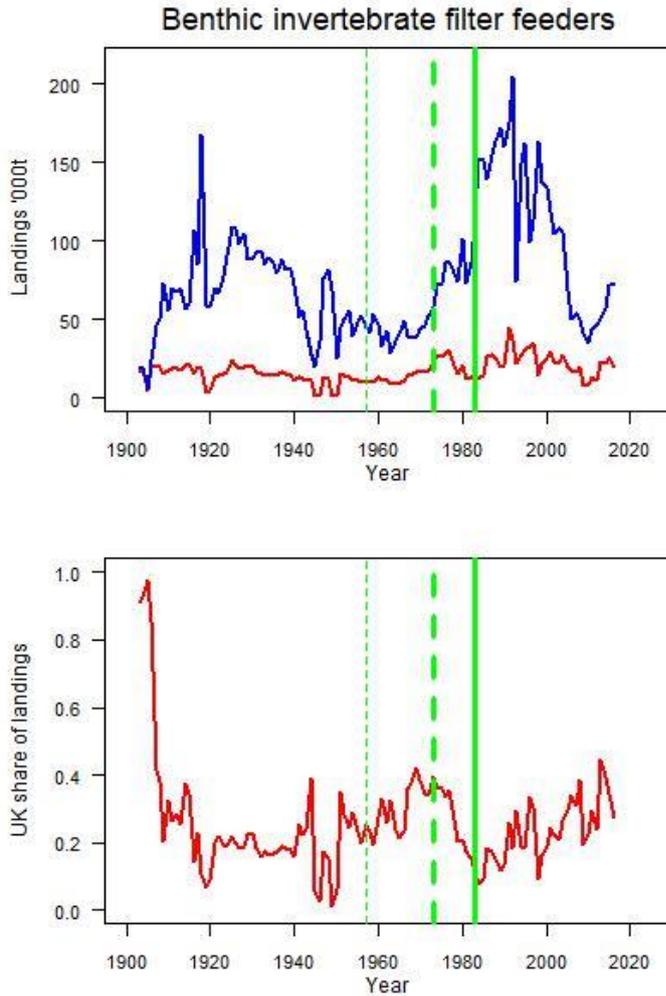


Figure A2.10. Landings history of all filter feeding shellfish species combined (e.g. molluscs such as scallops, mussels, oysters, clams) from the North Sea (FAO area 27.4). Details as in Figure A2.1, except that in this case we cannot estimate a Relative Stability fraction of the whole group since none are subject to EU CFP TACs and quotas. Similarly, of all species involved, only the Great Atlantic scallop is included in the STECF database so we cannot present a meaningful estimate of the proportion taken from UK waters.

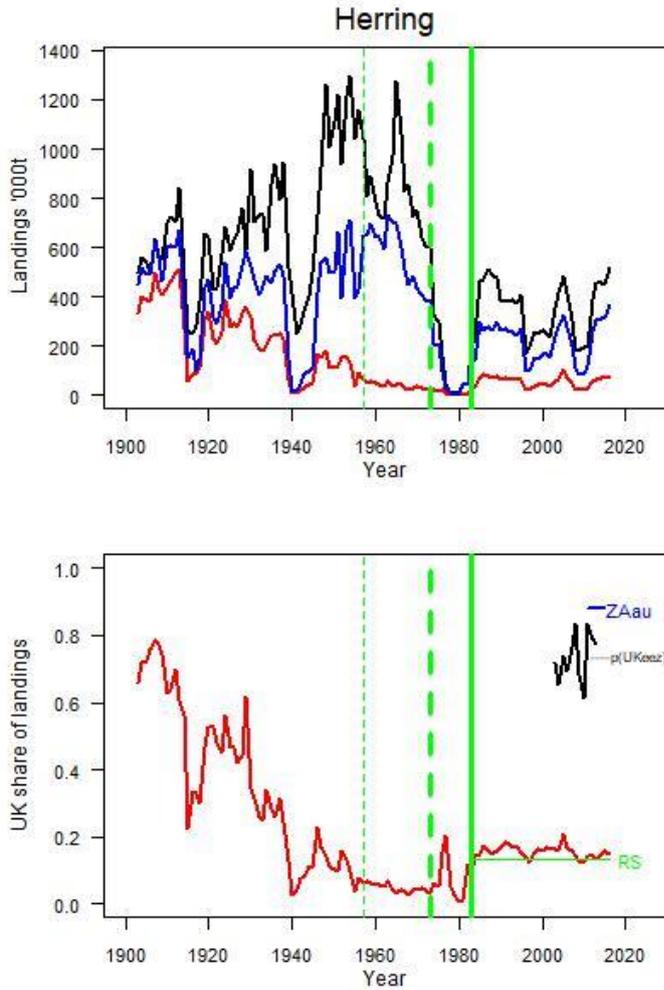


Figure A2.11. Landings history for herring in the North Sea. Details as in Figure A2.1. In the lower panel the Relative Stability fraction of TAC assigned to the UK is shown as a green horizontal line from 1983 onwards. The short blue line spanning 2011-2015 and labelled “ZAau” indicates the zonal attachment estimate for herring from the University of Aberdeen report.

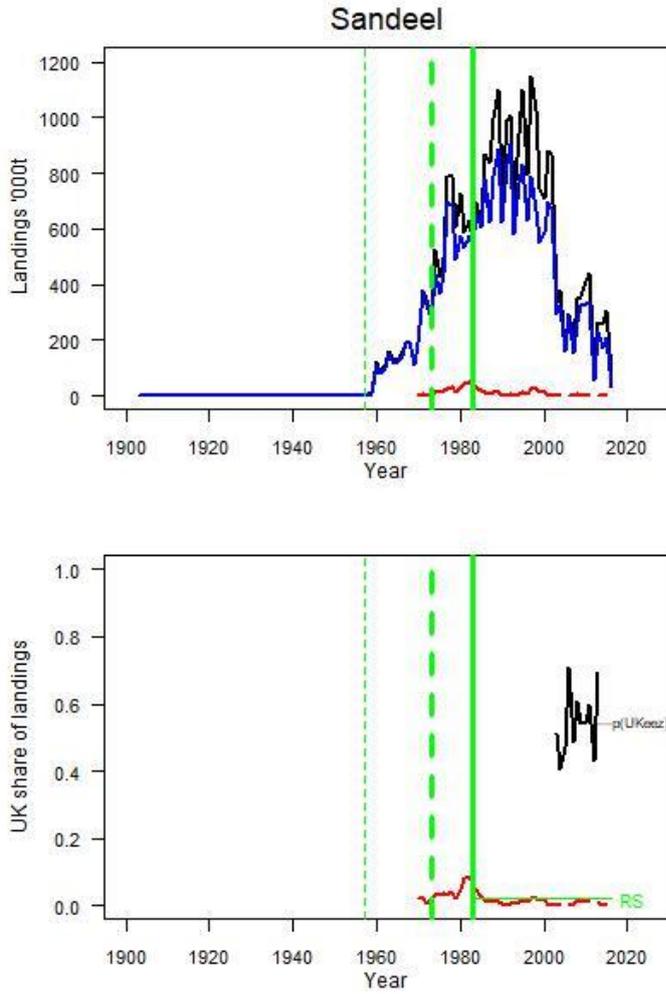


Figure A2.12. Landings history for sandeel in the North Sea. Details as in Figure A2.11. There are no biomass-based zonal attachment estimates available for sandeel.

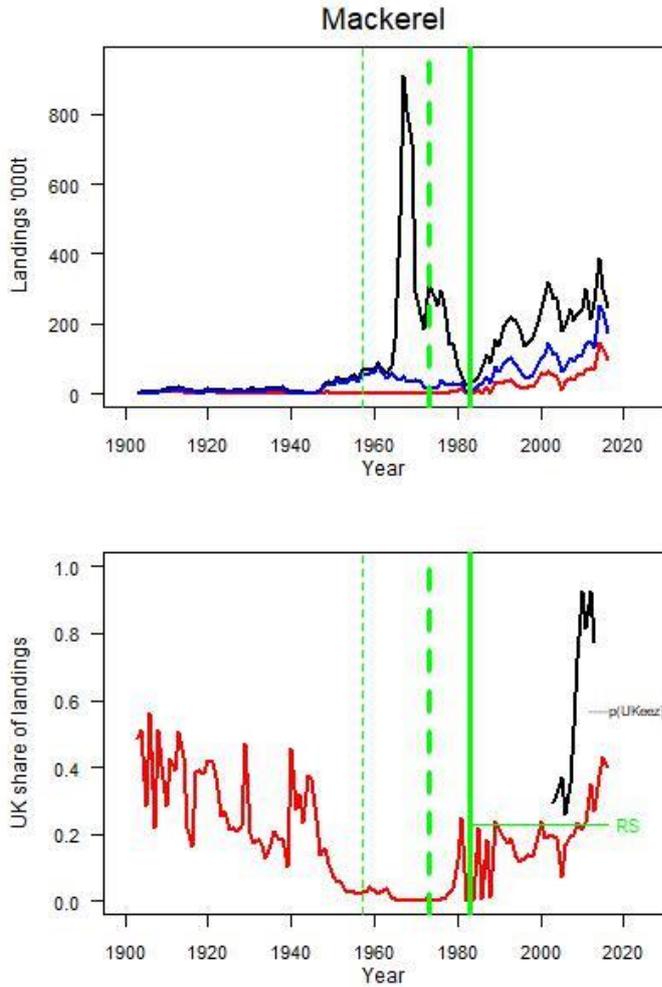


Figure A2.13. Landings history for mackerel in the North Sea. Details as in Figure A2.11. There are no biomass-based zonal attachment estimates available for mackerel in the North Sea.

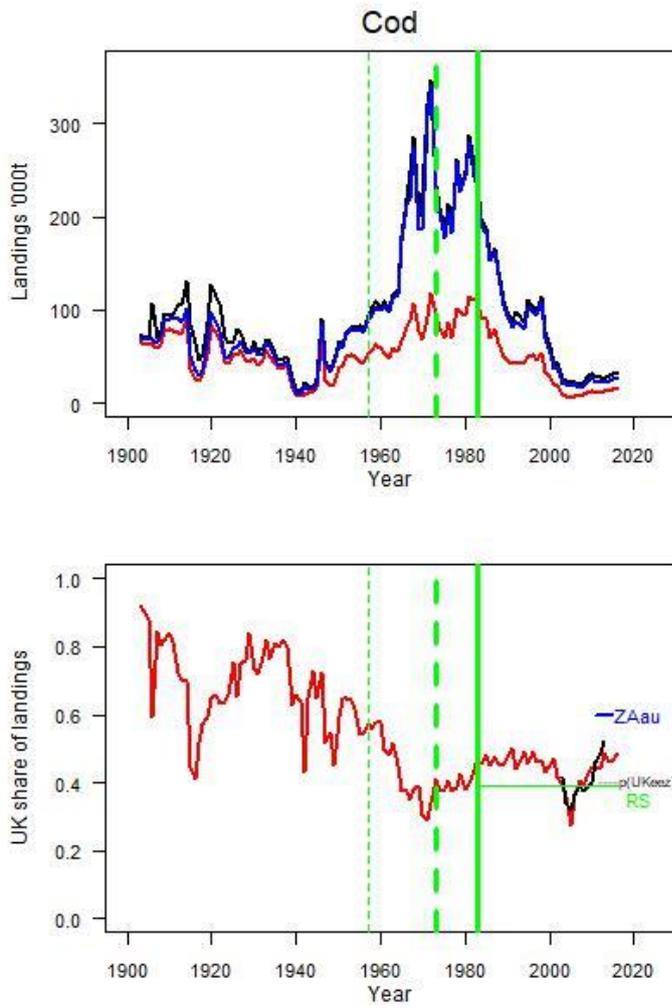


Figure A2.14. Landings history for cod in the North Sea. Details as in Figure A2.11.

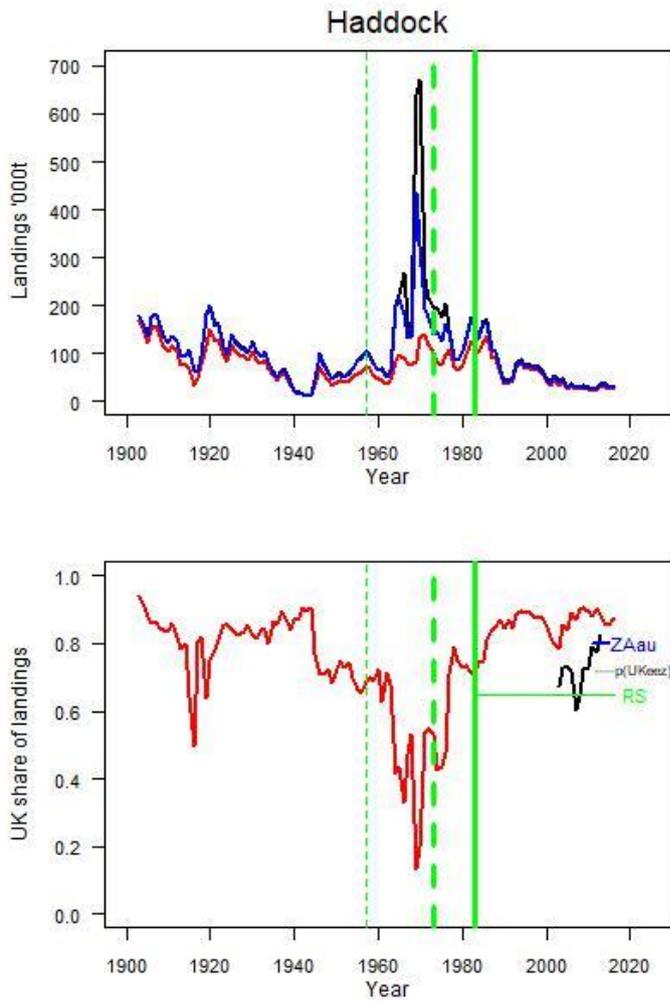


Figure A2.15. Landings history for haddock in the North Sea. Details as in Figure A2.11.

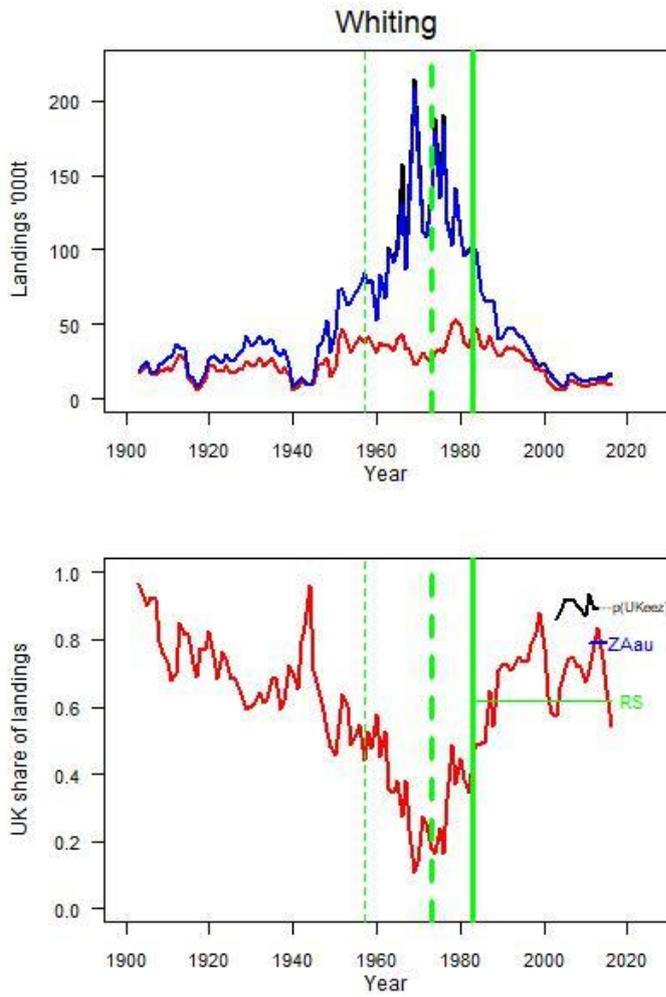


Figure A2.16. Landings history for whiting in the North Sea. Details as in Figure A2.11.

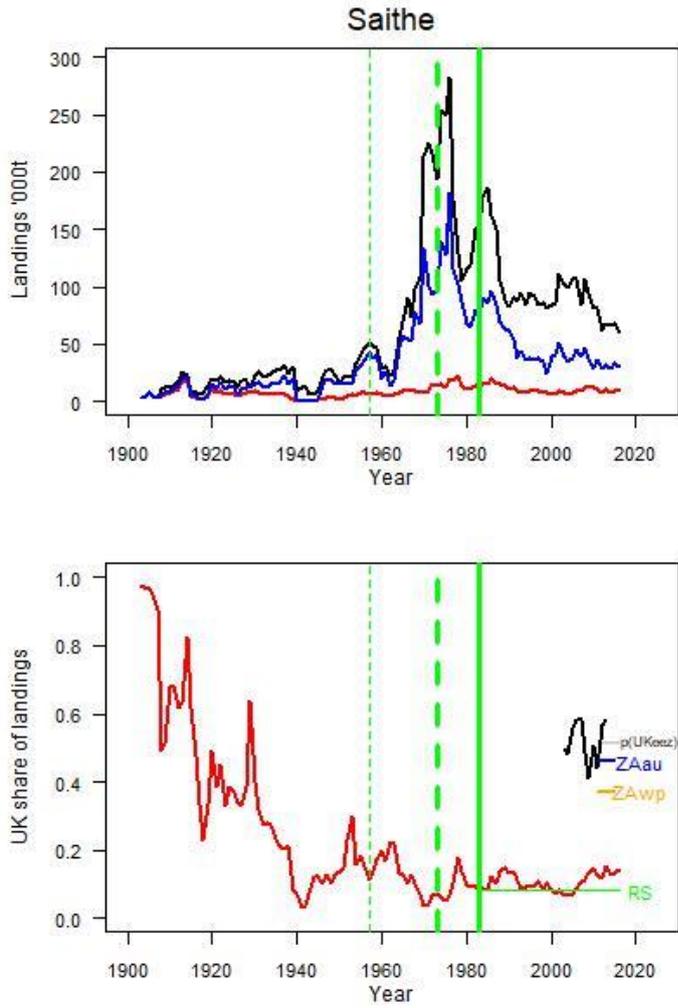


Figure A2.17. Landings history for saithe in the North Sea. Details as in Figure A2.11, except that in this case we also have an additional estimate of zonal attachment from the White Paper (shown in orange).

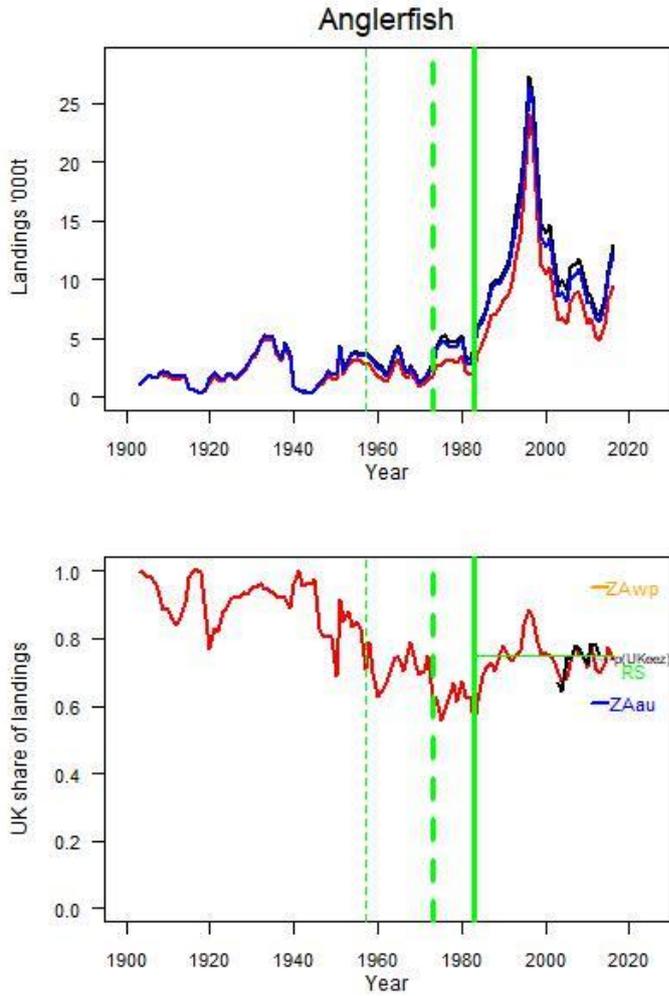


Figure A2.18. Landings history for anglerfish in the North Sea. Details as in Figure A2.11, except that in this case we also have an additional estimate of zonal attachment from the White Paper (shown in orange).

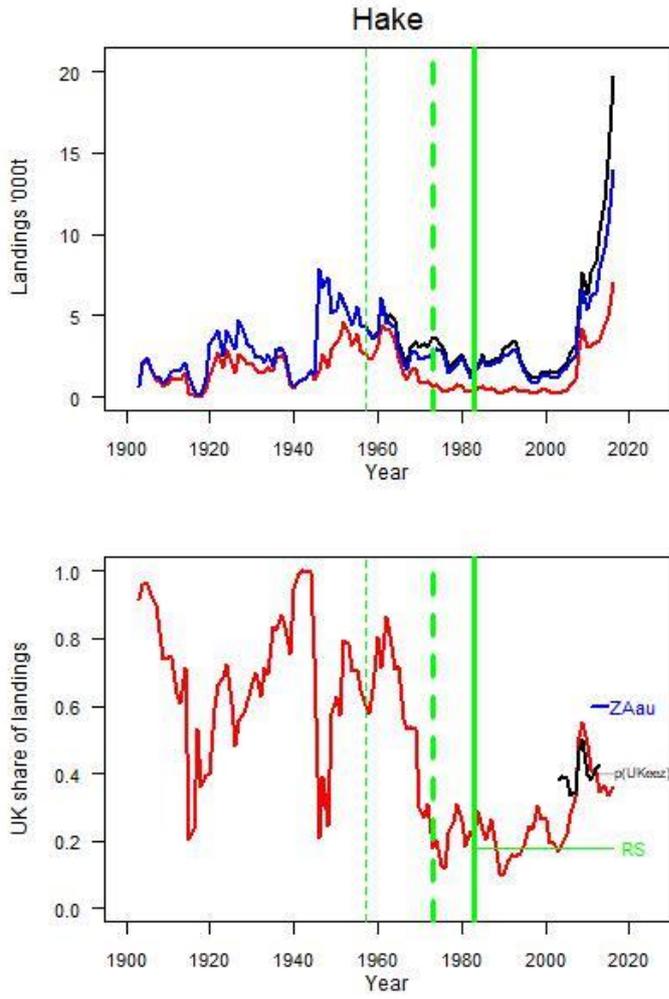


Figure A2.19. Landings history for hake in the North Sea. Details as in Figure A2.11

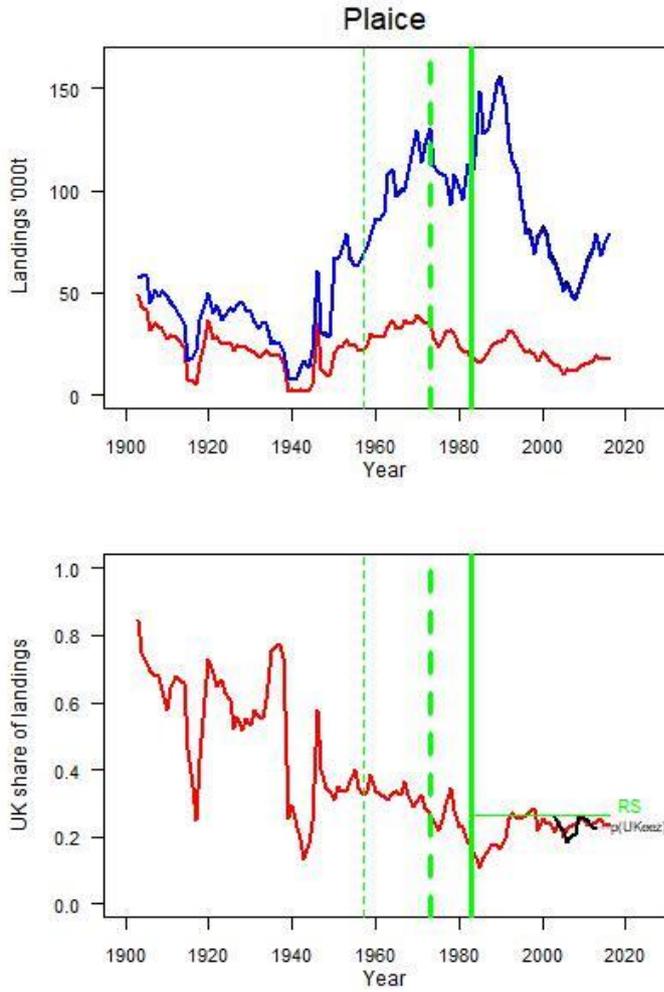


Figure A2.20. Landings history for plaice in the North Sea. Details as in Figure A2.11, except that there are no zonal attachment estimates for this species.

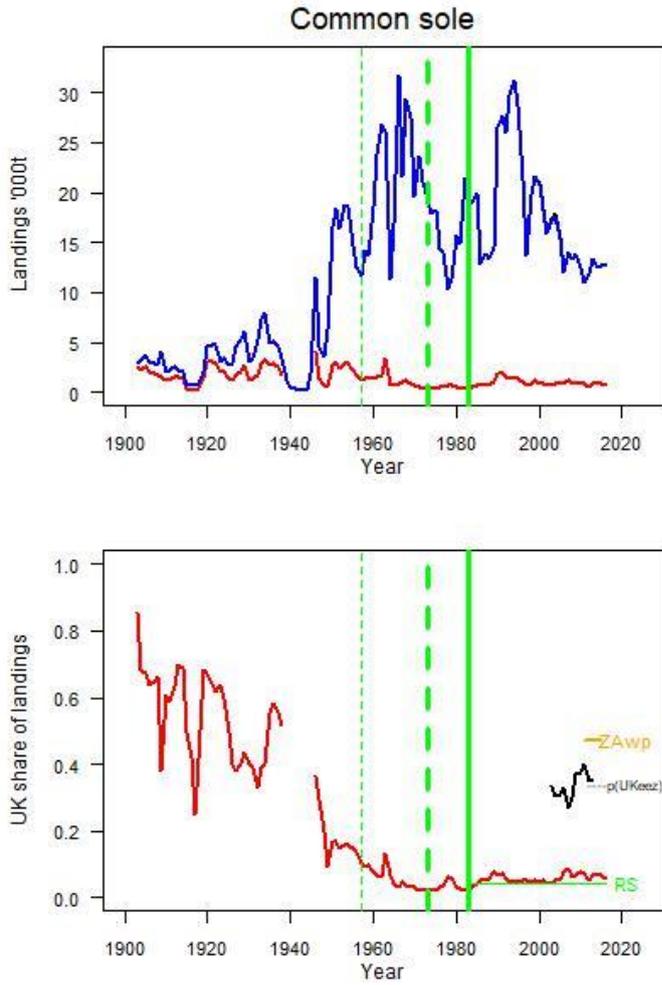


Figure A2.21. Landings history for common sole in the North Sea. Details as in Figure A2.11, except that the only zonal attachment estimate is from the White Paper.

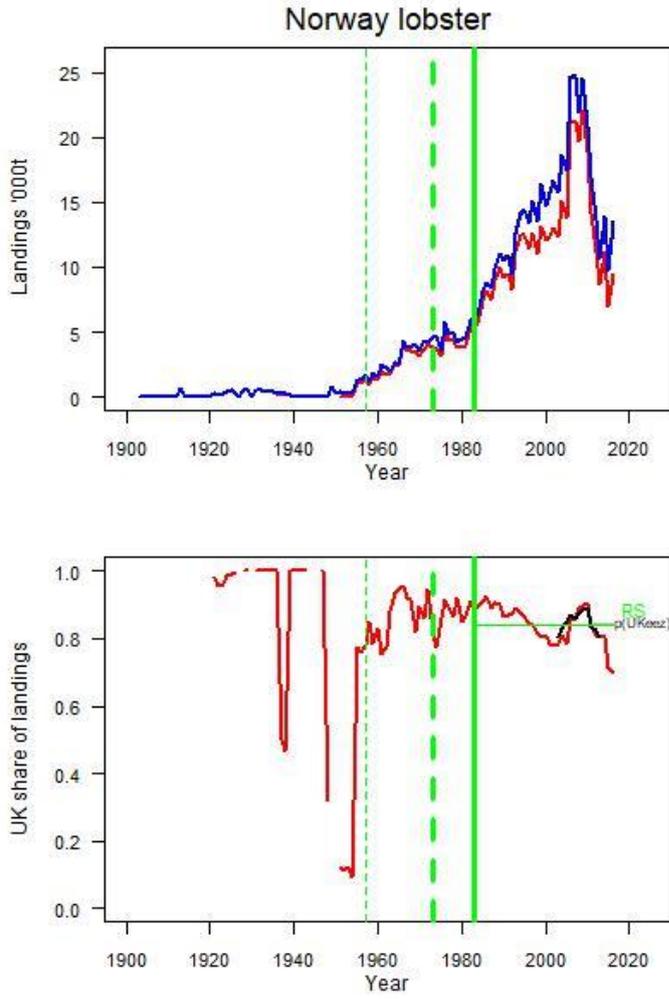


Figure A2.22. Landings history for Norway lobster in the North Sea. Details as in Figure A2.11, except that there are no zonal attachment estimates for this species.

APPENDIX 3. Stock projection models

Population projection model

For all the stocks concerned a standard age structured population model is used for projection summarised in the Table A3.1. Uncertainty in the projected populations was incorporated by considering measurement error in the starting populations and process error in recruitment and fishing mortality as outlined in Table A3.2. In addition to process error in recruitment (i.e. variability around the stock recruitment curve) uncertainty was considered in the structure of the stock-recruitment curve itself. This was done by drawing the two Beverton-Holt parameters from posterior distributions after fitting the curve to data.

Table A3.1. Population model equations

| | | |
|------|--|---|
| T1.1 | $N_{a,y} = N_{a-1,y-1} e^{-Z_{a-1,y-1}}$ | The population N at age a and year y decays exponentially with total mortality Z . |
| T1.2 | $Z_a = M_a + F_a$ | The total mortality Z is partitioned between natural mortality M , and fishing mortality F . |
| T1.3 | $F_a = \sum f_{a,k}$ | The total fishing mortality is the sum of the fleet fishing mortalities, $f_{a,k}$ where k indexes fleet. |
| T1.4 | $N_{1,y} = \alpha S_{y-1} / (1 + \beta S_{y-1})$ | The stock-recruitment follows a Beverton-Holt curve where S is the spawning stock biomass and the age of recruitment is 1. |
| T1.5 | $C_{a,y,k} = \frac{f_{a,y,k}}{Z_{a,y}} N_{a,y} (1 - e^{-Z_{a,y}})$ | The fleet catch, C , is calculated using the Baranov equation. |
| T1.6 | $S_y = \sum N_{a,y} mat_{a,y} w_{a,y}^s$ | Spawning stock biomass is the sum of the mature biomass at each age, where mat is the proportion mature and w^s is the stock weight at age. |
| T1.7 | $Y_{y,k} = \sum C_{a,y,k} w_{a,y}^c$ | Catch biomass is the sum of the biomass at each age, where w^c is the catch weight at age. |

Table A3.2. Stochastic variation introduced to the population model.

| | | |
|------|---|---|
| T2.1 | $N'_{a,1} = N_{a,y} \epsilon_a^s \epsilon^f$ | The population in the first year N' is derived by disturbing the nominal population, N , by a lognormal age specific error, ϵ_a^s , and a lognormal year specific error, ϵ^f , each with zero mean and standard deviations, σ^s and σ^f . |
| T2.2 | $f_{a,y} = f_{a,y-1} \epsilon_{a,y}^s \epsilon_y^f$ | Fishing mortality follows a random walk with process error, ϵ_y^f . An age specific error, $\epsilon_{a,y}^s$, changes the selection pattern annually. The errors, ϵ , are drawn from log normal distributions with zero mean and standard deviations, δ^s and δ^f . |
| T2.3 | $N'_{1,y} = N_{1,y} \epsilon_y^r$ | Recruitment is disturbed with noise, ϵ_y^r , drawn from a lognormal distribution with zero mean and standard deviation, σ^r . |

In the case of recruitment, the value of σ^r was estimated when fitting the Beverton-Holt equation (T1.4) to stock-recruitment data obtained from ICES stock assessments. For all the other error distributions values were assumed and are listed in Table A3.3.

Table A3.3. Error distributions used in projections

| Error S.D | Value | Description |
|------------|-------|--|
| σ^s | 0.1 | Lognormal measurement error on each N at age in year 1 |
| σ^f | 0.1 | Lognormal measurement error scaling N at all ages in year 1 |
| δ^s | 0.005 | Lognormal process error changing selectivity at each age each year |
| δ^f | 0.05 | Lognormal process error changing overall fishing mortality each year |

Data

The nominal population numbers at age for each stock were taken from the most recent ICES stock assessment report [3.1, 3.2]. These gave a population vector for a base year of 2017. Vectors of weights at age, fishing mortality at age, natural mortality and maturity were

obtained from the same source using tabulated values that ICES applies in the catch forecast. These are generally an average of recent years. Stock recruitment data were also obtained from the same ICES assessment reports.

ICES assesses stocks are often defined over a larger area than the North Sea alone. We have used the ICES stock boundaries as the basis for projections even though these include TAC components outside the North Sea proper. For most stocks the bias from ignoring these additional areas (usually the Channel and Kattegat) in the Brexit scenario will be very small. For saithe and haddock the inclusion of Division 6a (west of Scotland) may result in larger bias. This problem has to be traded against the errors introduced by attempting to partition out the part of the stock not in the North Sea which itself requires some arbitrary assumptions. Given that the contribution of 6a to the total biomass is minor for both haddock and saithe we considered it preferable to use the ICES boundaries.

Methods

The model was run for a projection period of 100 years using 2017 as the base year. Populations from year 50 onwards were considered to be at equilibrium for the purposes of calculating steady state mean yield and biomass. Yield was calculated only for human consumption landings with discards or industrial bycatch partitioned out of the total catch.

Stock-recruitment parameters were calculated by fitting equation T1.4 to the available ICES stock data using a Bayesian procedure [3.3] in order to obtain posterior distributions of the model parameters. Uniform priors were used for all model parameters. After fitting the model, 8000 simulated values from the posterior distributions for the parameters were saved for use in the projections.

When estimating MSY reference points, ICES uses only recent values of recruitment in the stock-recruitment function for some stocks. In this study we used the same base year for recruitment as use by ICES and these are given in Table A3.4

Table A3.4. Configuration values for each stock used in simulations

| Stock | Start year for recruitment data used to fit B&H model | Δ , the increment in F for post-Brexit scenario |
|---------|---|--|
| Cod | 1988 | 0.158 |
| Haddock | 2000 | 0.0 |
| Whiting | 1983 | 0.143 |
| Saithe | 2003 | 0.370 |
| Plaice | 1958 | 0.0 |
| Sole | 1957 | 0.258 |
| Herring | 2002 | 0.718 |

The projection procedure was as follows:

1. Draw an initial population using equation T2.1
2. Draw a pair of stock recruitment parameters and associated residual variance for use in the projection
3. Project the population one year ahead using the given initial fishing mortality
4. Calculate the current S (T1.6) and calculate recruitment using equation T1.4. Disturb the recruitment value using the residual variance from the fitted stock-recruitment model (T2.3).
5. Calculate human consumption landings (T1.7)
6. Update fishing mortality using equation T2.2
7. Repeat steps 3-5 until 100 years is reached.
8. Repeat 1-7 for 8000 replicates in order to simulate different starting populations and recruitment models.

Simulations were run assuming status quo fishing mortality, F_{sq} , (i.e. the 2017 F from the ICES assessment) as the baseline. Alternative scenarios were derived from the general assumption that in the absence of an agreement on shares of the stock TAC, the UK and the EU would unilaterally take a share they consider justified. In such a case, the proportions of the TAC taken by each party may sum to more than 1. In the case of the UK invoking a larger share without the agreement of other coastal states this implies that the revised fishing mortality, \tilde{F} , can be approximated by:

$$\tilde{F} = F_{sq}(1 - \Delta)$$

Where Δ is the difference between the UK current share (including quota swaps) under the CFP and the share asserted by the UK (see main report Table 1). This also assumes that the EU continues to assert its share as specified within the CFP.

Results from the simulations were summarised for both the short term (up to five years) and at equilibrium. As all ICES stocks have defined biomass reference points, the probability of exceeding the precautionary biomass threshold, B_{pa} , (i.e. the stock falls below this value) was also estimated. In addition, the expected human consumption landings and spawning stock biomasses were calculated.

In addition to the standard projections we ran sensitivity runs on herring and cod using a Ricker recruitment assumption ($N_{1,y} = \alpha S_{y-1} e^{\beta S_{y-1}}$) based on the full time series of available recruitment observations.

Derivation of the approximation used to calculate fishing mortality in the Brexit scenario.

For a given catch, C, the proportions by nation, p_i , sum to 1 , i.e.

$$C = Cp_1 + Cp_2 + Cp_3 + Cp_4 + \dots = C \sum_{i=1}^n p_i$$

Suppose that country A increases its proportion, p_A , by a value Δ , i.e.

$$p'_A = p_A + \Delta$$

The catch in this case is increased so that:

$$C' = C \sum p_i + \Delta C = C(1 + \Delta)$$

For the nominal catch, C , the associated fishing mortality, F , can be approximated by solving the catch equation:

$$(1) C = FB(1 - \exp(-z))/z$$

Where B is the fishable biomass. The revised catch, C' , due to increasing the share of country A with associate fishing mortality, F' , is given by:

$$(2) C' = C(1 + \Delta) = F'B(1 - \exp(-z'))/z'$$

Dividing (2) by (1) leads to:

$$(3) 1 + \Delta = \frac{F'z(1 - \exp(-z'))}{Fz'(1 - \exp(-z))}$$

Hence F' can be estimated by solving (3) given values for F , natural mortality M , p_A and Δ .

Note that $1 - \exp(-z) \approx z$ for small values of z , and hence:

$$z(1 - \exp(-z')) \approx z'(1 - \exp(-z))$$

So that (3) can be rearranged to:

$$(4) F' \approx F(1 + \Delta)$$

References for Appendix 3

[3.1] ICES 2018a. Report of the Herring Assessment Working Group for the Area South of 62°N (HAWG). ICES CM 2018/ACOM:07

[3.2] ICES 2018b. Report of the Working Group on Assessment of Demersal Stocks in the North Sea and Skagerrak (2018) [Provisional report].

[3.3] Lunn, D.J., Thomas, A., Best, N. & Spiegelhalter, D. 2000. WinBUGS — a Bayesian modelling framework: concepts, structure, and extensibility. *Statistics and Computing*, **10**, 325–337.

Summary outputs from the projection model showing the stock recruitment model, distributions of biomass and probabilities of falling below Bpa.

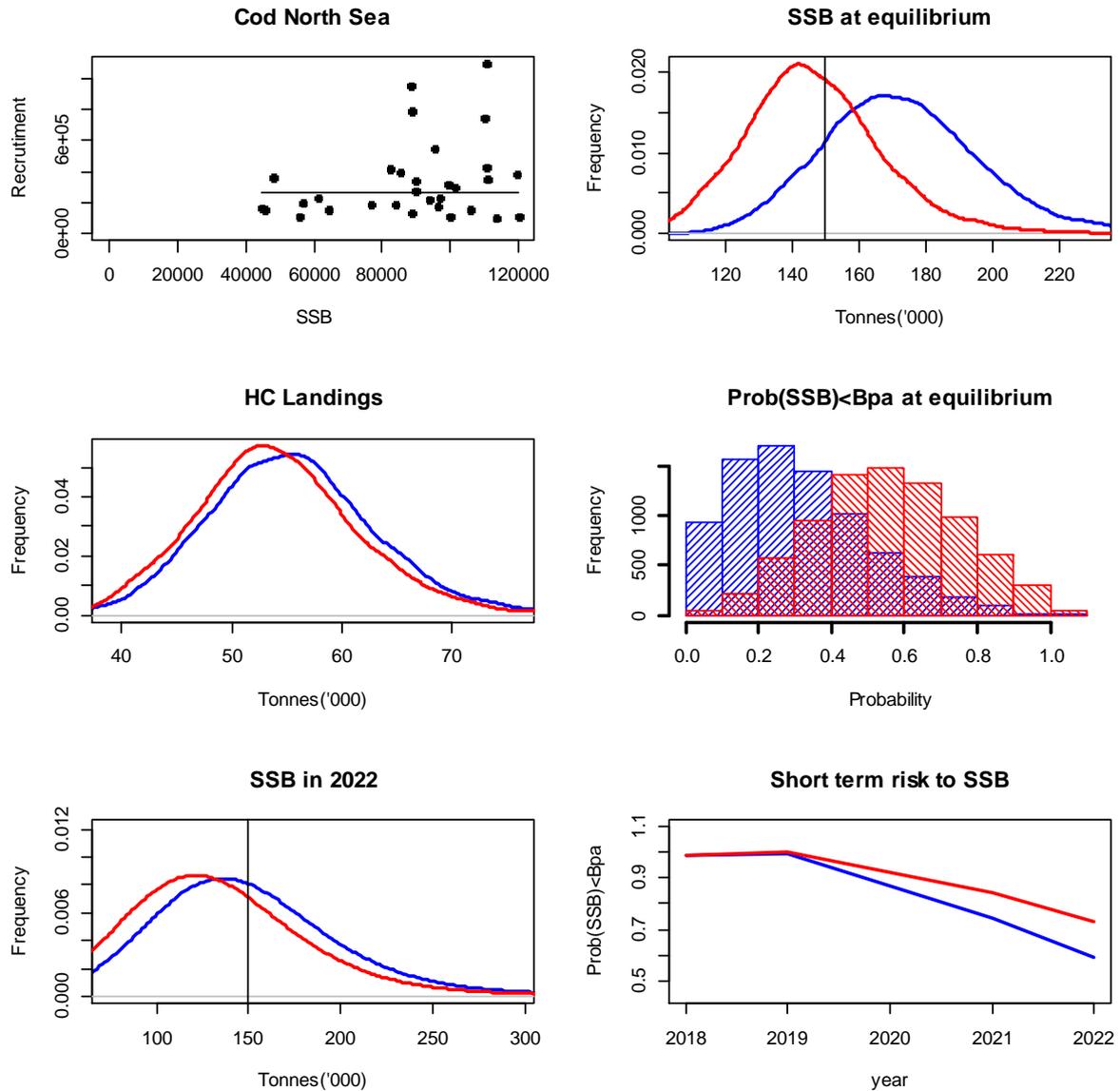


Figure A2.1. Cod. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

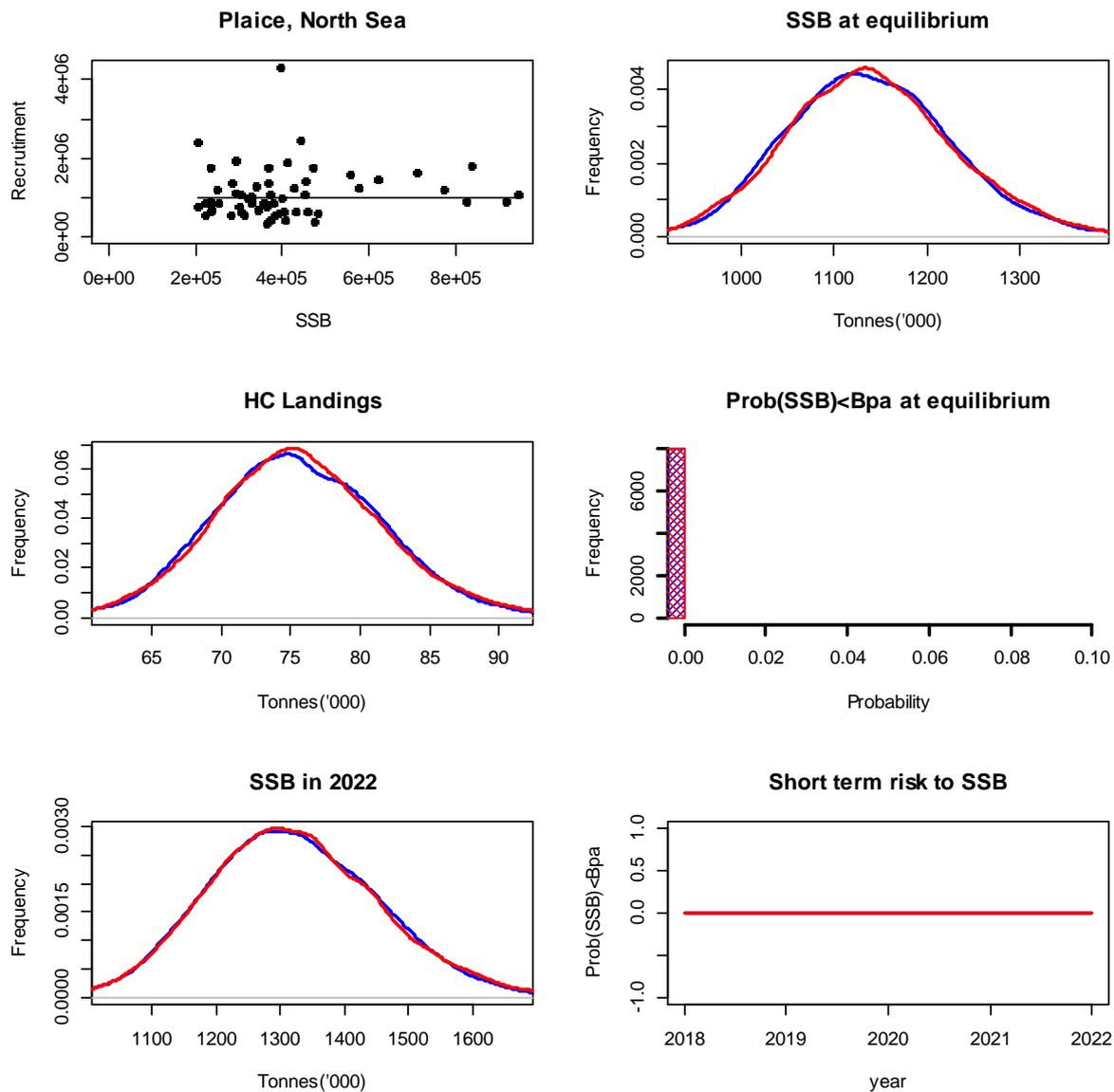


Figure A3.2. Plaice. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

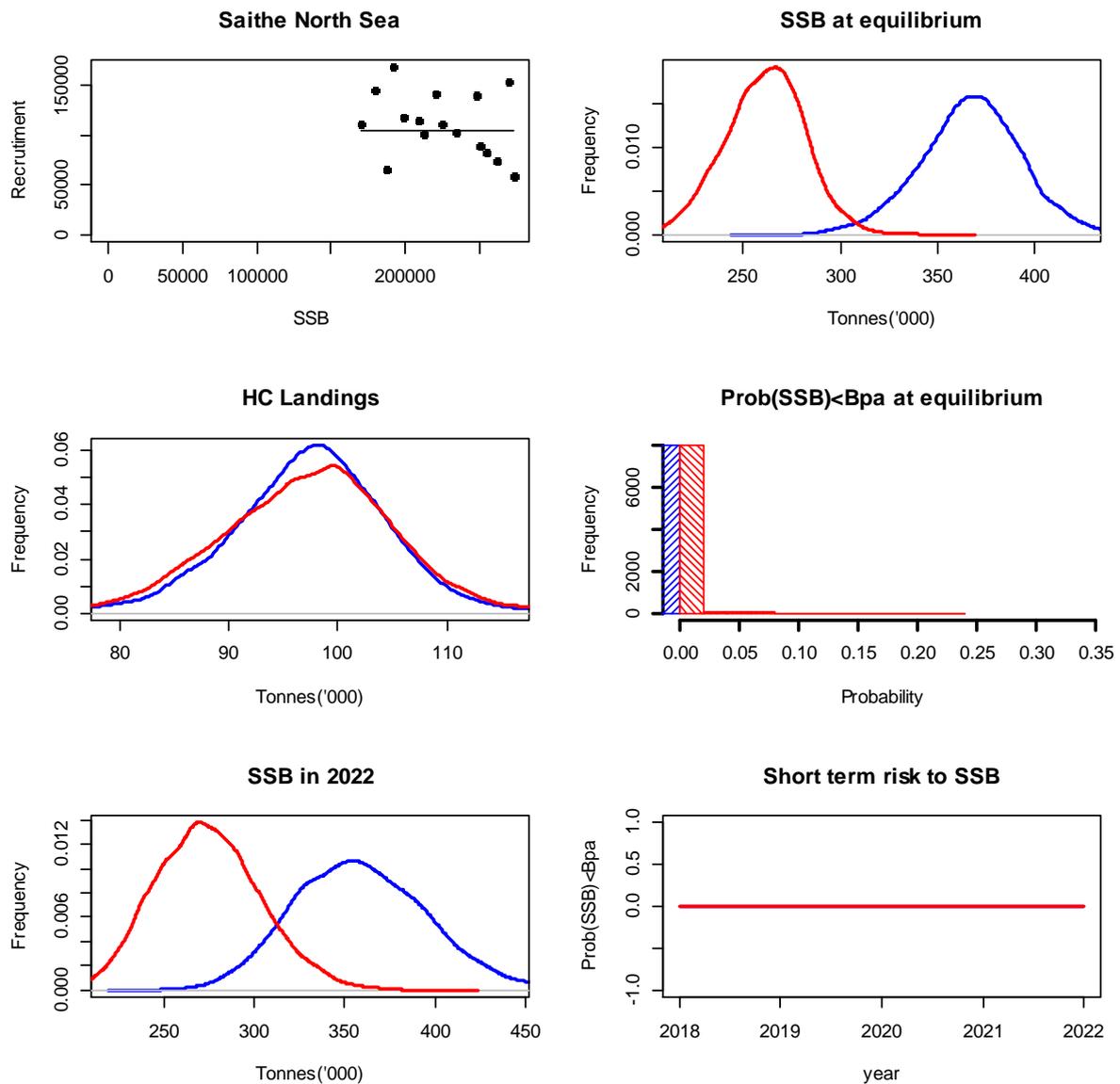


Figure A3.3. Saithe. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

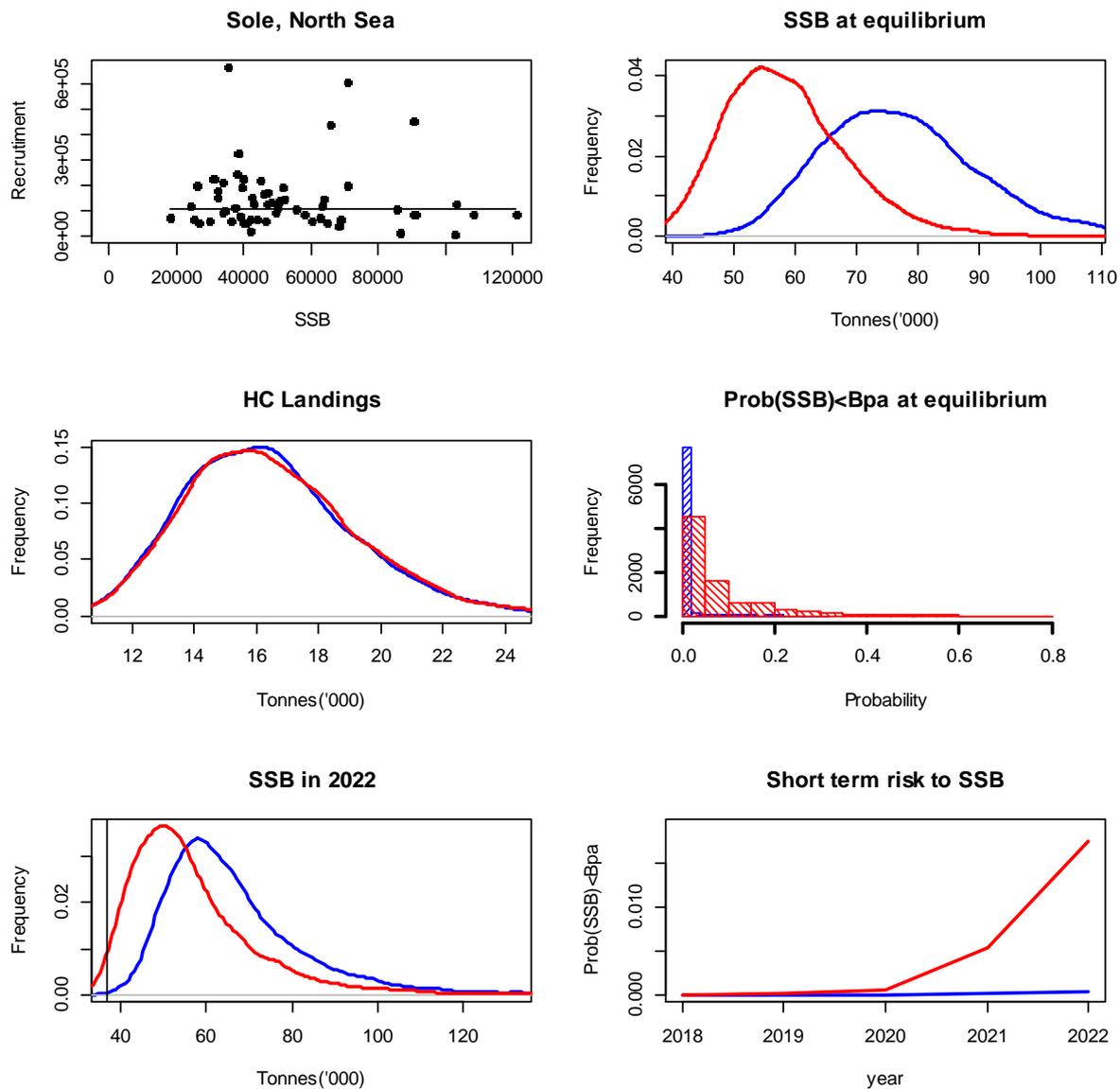


Figure A3.4. Sole. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

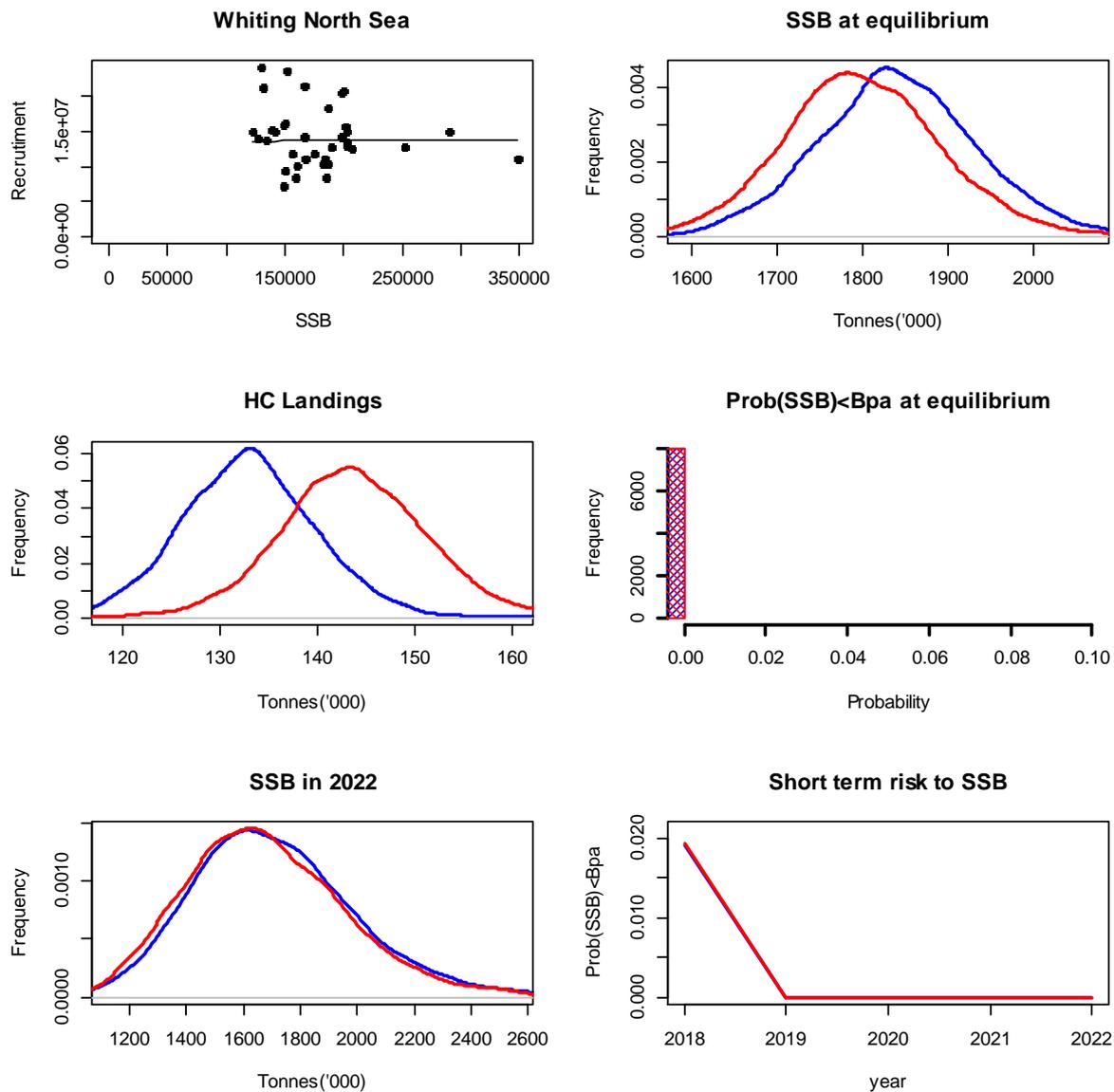


Figure A3.5. Whiting. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

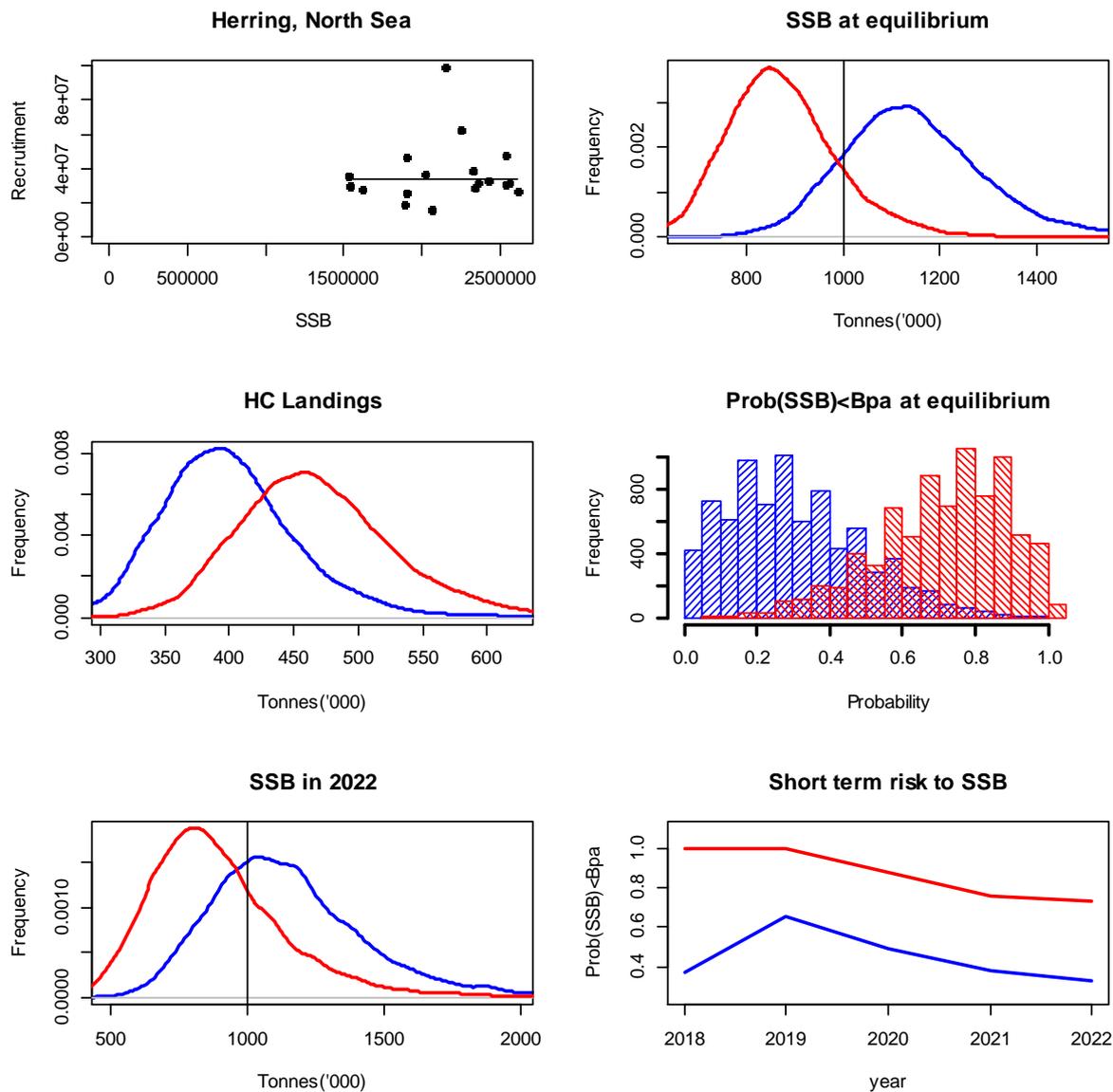


Figure A3.6. Herring. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

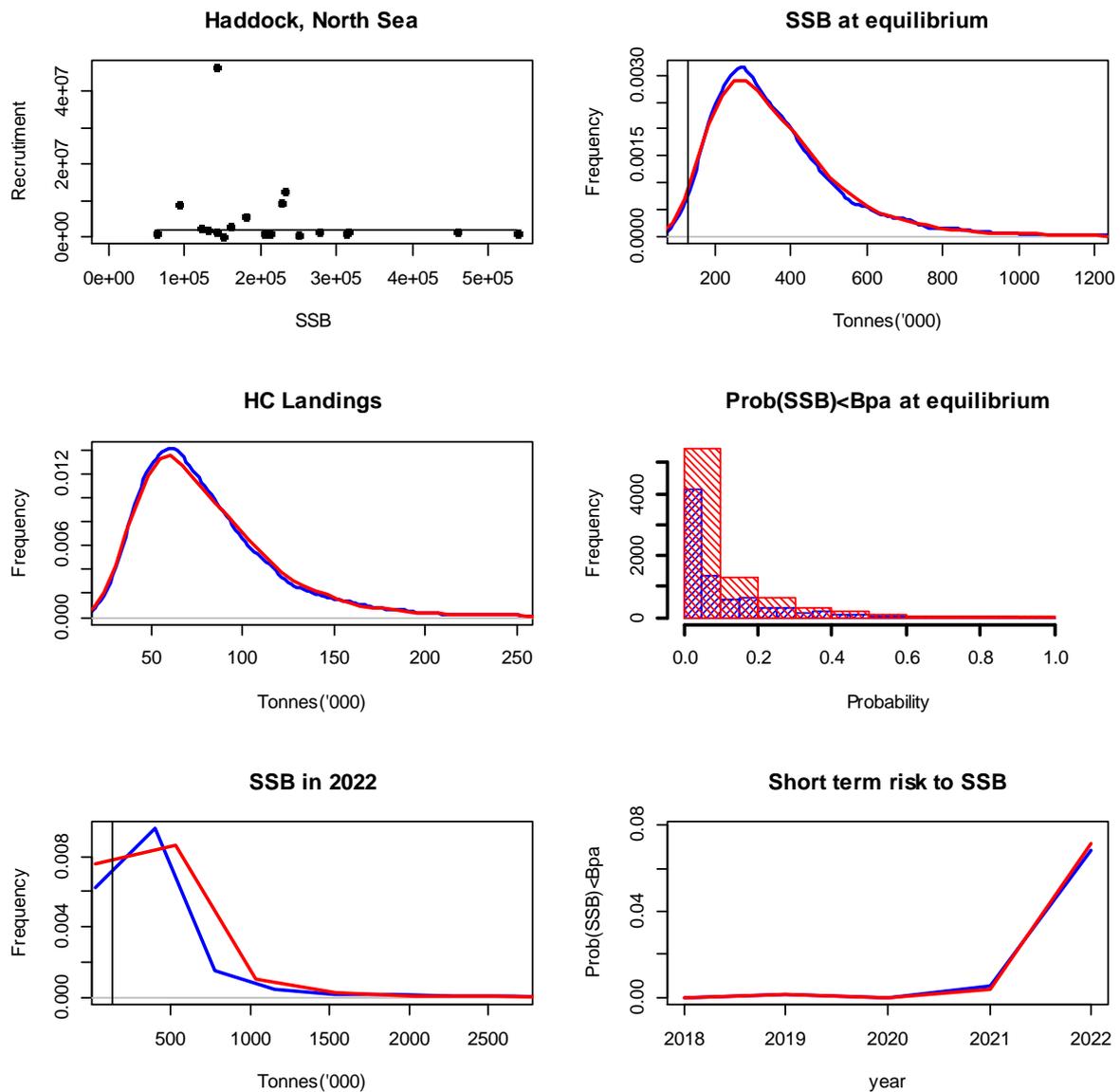


Figure A3.7. Haddock. Summary of stock projections. Blue lines show status quo fishing results and the red lines the result from the post-Brexit scenario. Distributions show the biomass and human consumption (HC) landings resulting from natural variability in recruitment, uncertainty in the stock recruitment model, uncertainty in the initial conditions and process variability in fishing mortality.

APPENDIX 4. Food web/ecosystem simulation model

The food web/ecosystem model used in this report is available as a software package for the R statistical environment (<https://gitlab.com/MarineResourceModelling/StrathE2E>). Version 2.0.0 was used to generate the results presented here.

The model as a whole comprises two sub-models – an ecology model and a fishing fleet model.

Ecology model general description

The ecology model [4.1, 4.2] is a network of mass conserving coupled ordinary differential equations (ODEs) describing spatially averaged rates of change in state variables representing organic detritus, dissolved inorganic nutrient, and living biomass. To simplify the description we can think of the variables as being divided between two coupled sub-networks: a predator-prey network - the food web - and a nutrient recycling network. Between the two, all marine life-forms are explicitly or implicitly included, ranging from bacteria to birds and mammals, but aggregated into coarse groups or ‘guilds’ defined primarily by feeding characteristics and diet preferences. All state variables are expressed in terms of nitrogen mass, since this element is the most commonly limiting in temperate shelf seas.

Each ODE comprises a set of rate-of-change terms representing a variety of biological and physical processes (Box A4.1). Biological terms describe the balance between gains due to assimilation of food, and losses due to mortality and metabolism. Some components of the food web are resolved into life-stages and for these the equations also include the balance between gains due to recruitment and losses due to developmental progression or spawning. In addition, all components of the model are replicated across homogeneous spatial compartments, so each ODE also includes terms representing advection, mixing and migration flows through the system.

The spatial structure is highly stylised, consistent with the coarse guild-definitions of the living and chemical components of the system. Two horizontally distinct bathymetric/hydrographic zones are distinguished – a shallow, vertically mixed zone influenced by tides and freshwater inputs, and a deeper, potentially seasonally stratified zone influenced by exchange with an external ocean (Figures A4.1, A4.2). For convenience here, we refer to these as the inshore and offshore zones respectively. The water column in the offshore/deep/seasonally stratified zone is divided vertically into two layers whilst the inshore/shallow/mixed zone is represented by a single layer. Seabed sediments are represented by up to three patches of different sediment habitat in each zone, each defined by median grain size and natural disturbance rates (Fig A4.1). State variables are resolved hierarchically to spatial compartments representing seabed habitats, water column layers and

bathymetric zones, with the largest (in terms of body size) and/or most mobile groups being represented at the coarsest spatial resolution (Table A4.1).

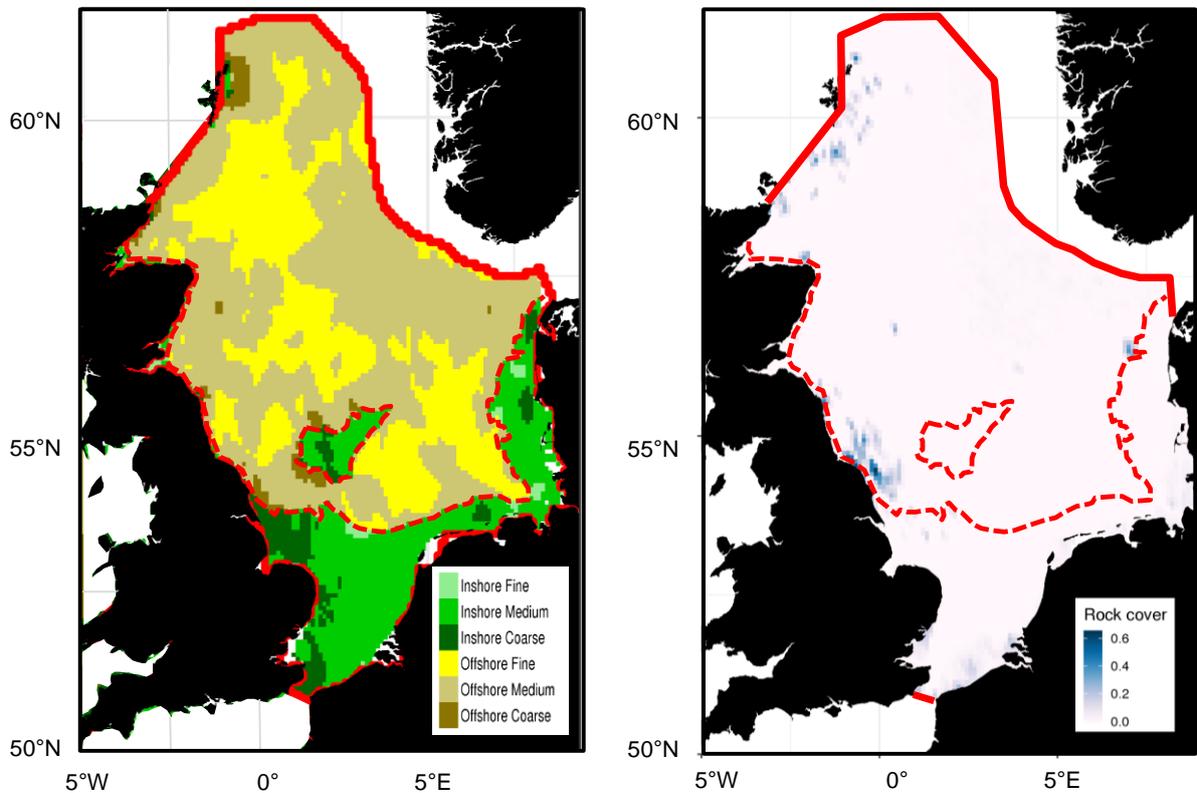


Figure A4.1. Map of StrathE2E North Sea model region. Within the North Sea the model resolves sub-area of seabed sediment habitat divided into inshore (shallower than 30m) and offshore. Within each depth zone, three sediment classes are represented – fine (muddy), medium (sandy) and coarse (gravel (left panel)). Within each of the six sediment habitats a proportion of the seabed area may present as exposed bedrock (right panel) which has different geochemical properties and in the inshore zone supports the kelp forests which are included in the model food web.

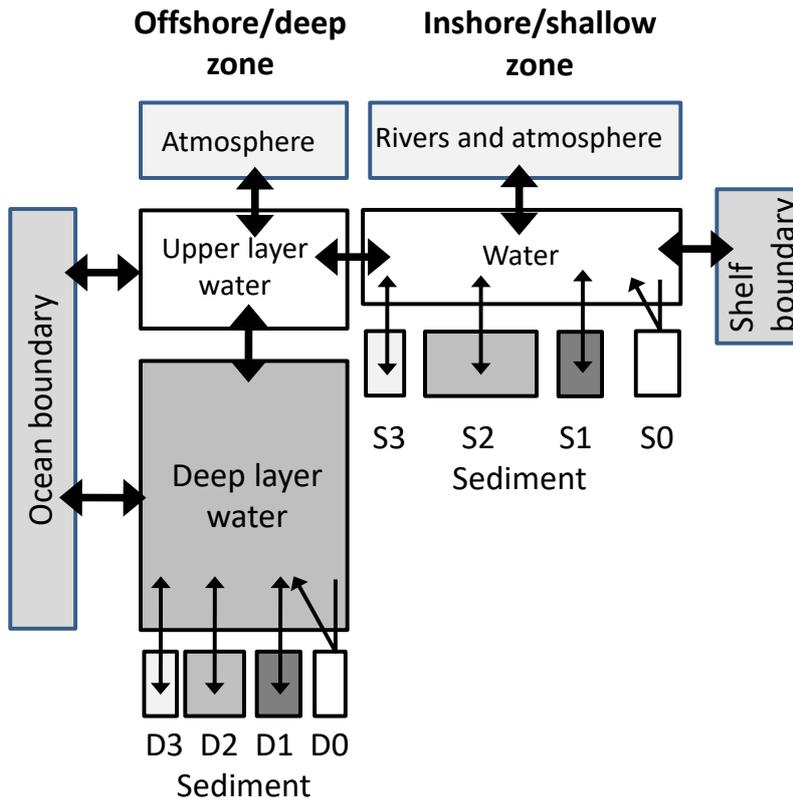


Figure A4.2. Schematic showing the spatial structure of the model, and how the inshore and offshore water column layers are connected to the external ocean environment and the seabed sediments. The sediment habitats S0 and D0 represent exposed bedrock which reflect settling material back into the water layers. The other sediment habitats absorb settling material and return nutrient to the water.

Box A4.1. Differential equations for the rate of change of living components in the StrathE2E model food web

General equation for the rate of change of a food web component (X) given a set of k prey types (N_k) and a set of j predator types (Y_j), is:

$$\frac{dX}{dt} = A \sum_k U_{X(N_k)} - \sum_j U_{Y_j(X)} - \varepsilon X - \delta X^2 + F_X - H(t)X - D(t)X + R_X$$

| | |
|----------------|---|
| $U_{v_1(v_2)}$ | Flux of ingestate to a predator (v_1) from prey (v_2). ($v_1, v_2 = X, N$ or $v_1, v_2 = Y, X$) |
| A | Assimilation efficiency. Ingestate not assimilated ($(1 - A) \sum_k U_{X(N_k)}$) is divided equally between a flux to dissolved ammonia, and a flux to detritus. |
| ε | Temperature dependent basal metabolic rate coefficient (generates a flux from body mass to ammonia) |
| δ | Density dependent mortality coefficient (generates a flux from body mass to a detritus category) |
| F_X | Integral of all vertical and horizontal advection and diffusion fluxes affecting the food web component |
| $H(t)$ | Harvest ratio (time-dependent rate of biomass capture by fisheries) |
| $D(t)$ | Time-dependent developmental export rate for the food web component X . For $X =$ adult stages, $D(t)X$ represents the flux of spawning products to the egg, larval and juvenile (ELJ) stage. For $X =$ ELJ stages, $D(t)X$ represents the settlement flux to adults. For food web components lacking demographic structure, $D(t) = 0$ |
| R_X | Recruitment flux to the food web component X . For $X =$ adult stages, R_X is equal to the settlement flux from the ELJ stage. For $X =$ ELJ stages R_X is equal to the flux of spawning products from the adults. For food web components lacking demographic structure, $R_X = 0$ |

General equation for the flux of ingestate to a predator (v_1) from prey (v_2) is:

$$U_{v_1(v_2)} = \frac{v_1 \cdot v_2 \cdot \rho_{v_1(v_2)} \cdot U_{max_{v_1}}}{v_2 + h_{v_1}}$$

| | |
|-------------------|---|
| $\rho_{v_1(v_2)}$ | Preference of the predator v_1 for the prey class v_2 . For a given predator class, the sum of the preference coefficients over all prey classes = 1. |
| $U_{max_{v_1}}$ | Temperature-dependent maximum uptake rate of the predator v_1 |
| h_{v_1} | Half-saturation constant for uptake of prey by the predator v_1 (temperature independent) |

For phytoplankton ($v_1 =$ phytoplankton ($X = P$)), the assimilation efficiency $A = 1$, temperature dependent basal metabolic rate coefficient $\varepsilon = 0$, and there is no demographic structure so $D(t) = 0$ (and hence $R_X = 0$). The uptake of prey ($v_2 =$ dissolved nutrient N_k) has a light-dependent term:

$$U_{P(N_k)} = \text{Min} \left\{ 1, \frac{L(t)}{L_{max}} \right\} \frac{P \cdot N_k \cdot \rho_{P(N_k)} \cdot U_{max_P}}{N_k + h_P}$$

| | |
|-----------|--|
| $L(t)$ | Time-dependent light intensity |
| L_{max} | Saturation light intensity for nutrient uptake |

For the top-predators in the food web (birds, seals and cetaceans), uptake of prey follows the predator-density dependent Beddington-DeAngelis function [4.3, 4.4] rather than Michelis Menten, with an additional parameter γ :

$$U_{v_1(v_2)} = \frac{v_1 \cdot v_2 \cdot \rho_{v_1(v_2)} \cdot U_{max_{v_1}}}{v_2 + h_{v_1} + \gamma v_1}$$

Table A4.1. Ecology model state variables and spatial hierarchy

| Differentiated by bathymetric zone and sediment habitat | Differentiated by bathymetric zone and water column layer | Differentiated by bathymetric zone with modelled vertical distribution | Differentiated by bathymetric zone only |
|--|--|---|---|
| Sediment bacteria and labile detritus | Nitrate | Omnivorous zooplankton | Suspension/deposit feeding benthos |
| Refractory sediment detritus | Ammonia | Carnivorous zooplankton | Carnivore/scavenge feeding benthos |
| Pore-water nitrate | Suspended bacteria and detritus | Larvae of suspension/deposit feeding benthos | Planktivorous fish |
| Pore-water ammonia | Phytoplankton | Larvae of carnivore/scavenge feeding benthos | Demersal fish (divided into fishery quota-limited and non-quota components) |
| Fishery discards | | Larvae of planktivorous fish | Migratory fish |
| Corpses | | Larvae of demersal fish | Pinnipeds |
| Macrophytes | | | Seabirds |
| | | | Cetaceans |

Table A4.2. Ecology model parameters, input and outputs

| |
|--|
| Static configuration data |
| Model domain sea surface area; area-proportions of bathymetric zones and water column layer thicknesses; area-proportions of seabed habitats and median grain sizes of sediments |
| Parameters for deriving sediment porosity, permeability and organic nitrogen content in each seabed habitat from median grain size, and light attenuation coefficients from suspended particulate matter (SPM) concentration |
| Ocean biomass of migratory fish stock and the annual proportion entering the model domain |
| Monthly resolution internal driving data |
| Proportion of each seabed habitat sediment layer volume disturbed by natural bed shear stress per unit time. |
| Vertical mixing and horizontal advection rates between compartments within the model |
| Temperature and suspended particulate matter concentrations in water column layers, sea surface irradiance in each depth zone, significant wave height adjacent to the coast |
| Monthly resolution external boundary influxes of nutrient |
| Volume inflows across the external ocean boundaries of the model and from rivers, and concentrations of nutrient, phytoplankton and suspended detritus in the inflows |
| Atmospheric deposition of nutrient to the sea surface |
| Inputs from the fishing fleet model |
| Bathymetric zone harvest ratios and discard rates for each resource group |
| Area-proportion of each seabed habitat abraded by trawling per unit time |
| Proportion of discards deposited over each seabed habitat |
| Biological parameters (* indicates fitted parameters) |
| *Prey preference parameters for each predator-prey pairing |
| *Maximum uptake rate and prey half saturation concentration for each consumer group |
| *First-order rate coefficients for microbial processes |
| *Density dependent mortality coefficients |
| *Coefficients for active horizontal migration rates of fish and top-predators |
| *Sinking rates for detritus |
| *Parameters for the exploitable fraction of biomass for each group subjected to fishing. |
| Saturating irradiances for nutrient uptake by phytoplankton, and carbon uptake by macrophytes |
| Assimilation efficiency for each consumer group |
| Maximum and minimum nitrogen:carbon ratios for macrophytes |
| Food-independent metabolic rates for each consumer group |
| Q ₁₀ temperature dependency coefficients for autotrophic and heterotrophic maximum uptake rates, metabolic rates and microbial processes |
| Annual weight-specific fecundities for fish and benthos groups; start and end dates for egg production, and for recruitment of larval stages to the settled stocks |
| Start and end dates for immigration and emigration of migratory fish |
| Parameters for relationship between demersal fish biomass and a) proportion of non-quota demersal fish and b) proportion of undersize quota-limited and non-quota fish in the catches |
| Model outputs (all at daily intervals) |
| Mass of each state variable |

| |
|--|
| Model import and export fluxes (transport, atmospheric deposition, river inflows, denitrification, fishery landings) |
| Derived internal fluxes: consumption flux for each prey-predator pair, consumption and production fluxes of nitrate and ammonia in each depth zone and layer, fishery discards and offal |

Representation of fisheries

Living biomass groups considered vulnerable to directed harvesting or incidental by-catch by fishing gears are the planktivorous, demersal and migratory fish; carnivorous/scavenge and filter/deposit feeding benthos; carnivorous zooplankton (squid); birds, seals and cetaceans (whales). Capture and either landing or discarding of these groups is represented as first-order rate processes in the relevant ODEs. Rate coefficients are the harvest rate (proportion of mass captured per unit time), and discard rate (proportion of catch weight returned to the sea as discards) for each element of a rectangular matrix of catchable resource groups and seabed habitats. These are external driving data for the ecology model, generated by the separate model of fishing fleets. The fraction of catch which is not discarded as whole fish or as offal as a result of processing at sea is treated as an export flux from the model (landings).

Three effects of sediment disturbance by bottom-contact gears are explicitly represented in the ecology model - release of pore-water nutrients, resuspension of sediment detritus, and damage mortality of benthos. The key property driving these processes is the area-proportion of each seabed sediment habitat abraded per unit time, which is generated by the fishing fleet model and delivered as an input to the ecology model [4.2].

Fleet model description

The fleet model, developed from a prototype by Heath *et al.* (2015) [4.2], is a static, matrix-based scheme. Key inputs are, for each gear type, the activity density and its proportional distribution across seabed habitat types, fishing power, and the seabed abrasion rate (Table A4.3). Activity density is defined as the deployment duration of a given gear per unit sea surface area in a given time interval, integrated across all vessels (units: m^{-2}). The power of a gear is a measure of its efficiency at catching biomass of a given resource group. The product of activity density and power is a quantity that we refer to as fishing effort. For a given resource group, effort is proportional to the harvest rate and can be summed across gears.

The fleet model also transfers to the ecology model, a set of parameters for relationships describing systematic changes in proportions by weight of non-target by-catch and undersize (less than legal minimum or marketable landing size) demersal fish in commercial catches, as a function of biomass in the sea. The default parameters for these relationships are based on observational data from e.g. trawl survey data, and allow discard patterns to vary dynamically during a simulation rather than according to static parameters.

Table A4.3. Fishing fleet model inputs and outputs

| |
|--|
| Input data for each gear type (maximum 12 types) |
| Annual model domain averaged fleet activity density (number of boats x time spent fishing per boat, per day, per unit area) |
| Proportion of annual activity over each model seabed habitat |
| Selectivity (catching power) for each ecology model resource group |
| Discard rate for each ecology model resource group |
| Proportion of each catch group processed (gutted) at sea |
| Seabed area abraded per unit activity |
| |
| Gear-independent parameters |
| Parameters for scaling effort (activity x power) to harvest ratio for each ecology model resource group |
| Seabed sediment penetration depth (common value across all gears) |
| Damage-related mortality rate of benthos per bottom-contact gear pass (common value across all gears) |
| Proportion by weight of viscera for catch groups processed at sea |
| |
| Model outputs |
| Bathymetric zone harvest ratios and processing-at-sea and discard rates for each ecology model resource group due to all gears combined (required for input to the ecology model) |
| Area-proportion of each seabed habitat abraded per unit time by all gears combined (required for input to the ecology model) |
| Proportion of discards from all gears combined, which are deposited over each seabed habitat (required for input to the ecology model) |
| For each bathymetric zone separately, proportion of total effort directed at each ecology model resource group which is attributable to each gear (required for disaggregating simulated landings and discards to individual gears from ecology model output) |

Parameterisation of selectivity for top predators.

The parameters linking activity density (time spent fishing per km² per year) of each gear-group in the model and the resulting harvest ratio on each top-predator group (proportion of stock biomass caught per year) were established from a thorough survey of literature and national reports, analyses of by-catch and strandings data on birds, seals and cetaceans (e.g. [4.5]), and results from statistical modelling of line-survey data on seabirds-at-sea and cetacean abundances (pers.comm, Dr James Waggitt & Dr Peter Evans, Bangor University). Data on catches of Minke whales in the Norwegian sector of the North Sea were provided by the Norwegian Directorate of Fisheries (Table A4.4).

Table A4.4. Fishing gears for which there are quantitative data on by-catch weights of particular species of top-predators, together with North Sea-wide estimates of total harvest ratio (proportion of vulnerable stock caught per year) during 2003-2013. Further details to be provided in a separate report in preparation.

| Gear | Vulnerable seabird species | Vulnerable seal species | Vulnerable cetacean species | Seabird group harvest ratio | Seal group harvest ratio | Cetacean group harvest ratio |
|-------------------|--------------------------------------|--------------------------------|--|------------------------------------|---------------------------------|-------------------------------------|
| Demersal gillnets | Guillemot, razorbill, fulmar, gannet | Grey seal | Common dolphin, striped dolphin, harbour porpoise | 2.58×10^{-5} | 4.98×10^{-3} | 4.39×10^{-3} |
| Pelagic trawl | Gannet | | Common dolphin, bottlenose dolphin, striped dolphin, pilot whale | 1.47×10^{-4} | | 2.64×10^{-4} |
| Pelagic seine | Gannet | | Common dolphin | | | 3.07×10^{-5} |
| Pelagic longlines | Fulmar | | | 4.41×10^{-5} | | |
| Creels & pots | | | Fin whale, Minke whale | | | 3.04×10^{-4} |
| Norwegian whaler | | | Minke whale | | | 1.52×10^{-3} |

For each combination of gear and by-catch species, we assembled a scatter plot of the activity rate (effort, hours fished per km² per year) by a given gear and national segment of the fleet in a region, and the corresponding species harvest ratio (proportion of biomass captured per year). By aggregating across species, regions and nations, we produced scaling relationships between activity and harvest ratios for each ecological group (seabirds, seals, cetaceans). Examples of these scatterplots are shown in Figures A4.3-A4.5.

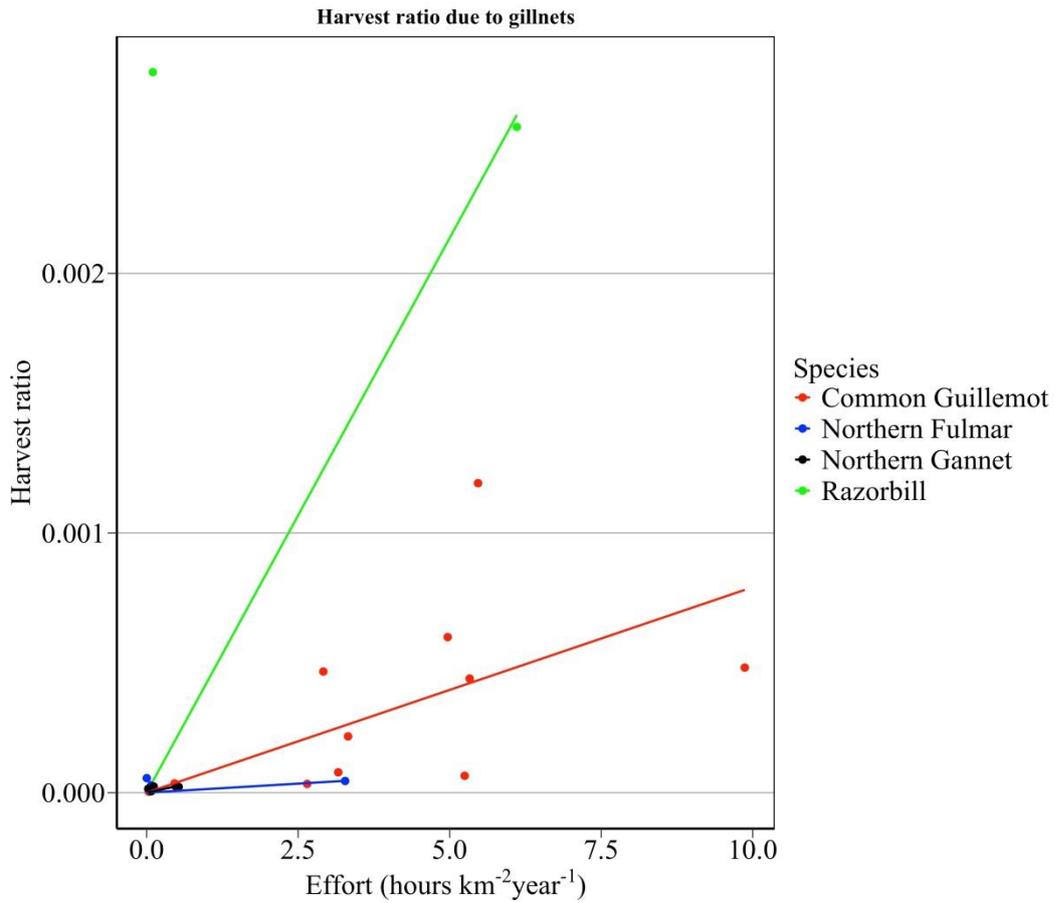


Figure A4.3. Scatterplots of the relationship between demersal gillnet effort and the harvest ratio (y^{-1}) on seabird species. For each species, each point represents an individual national assessment of by-catch and effort in a given region.

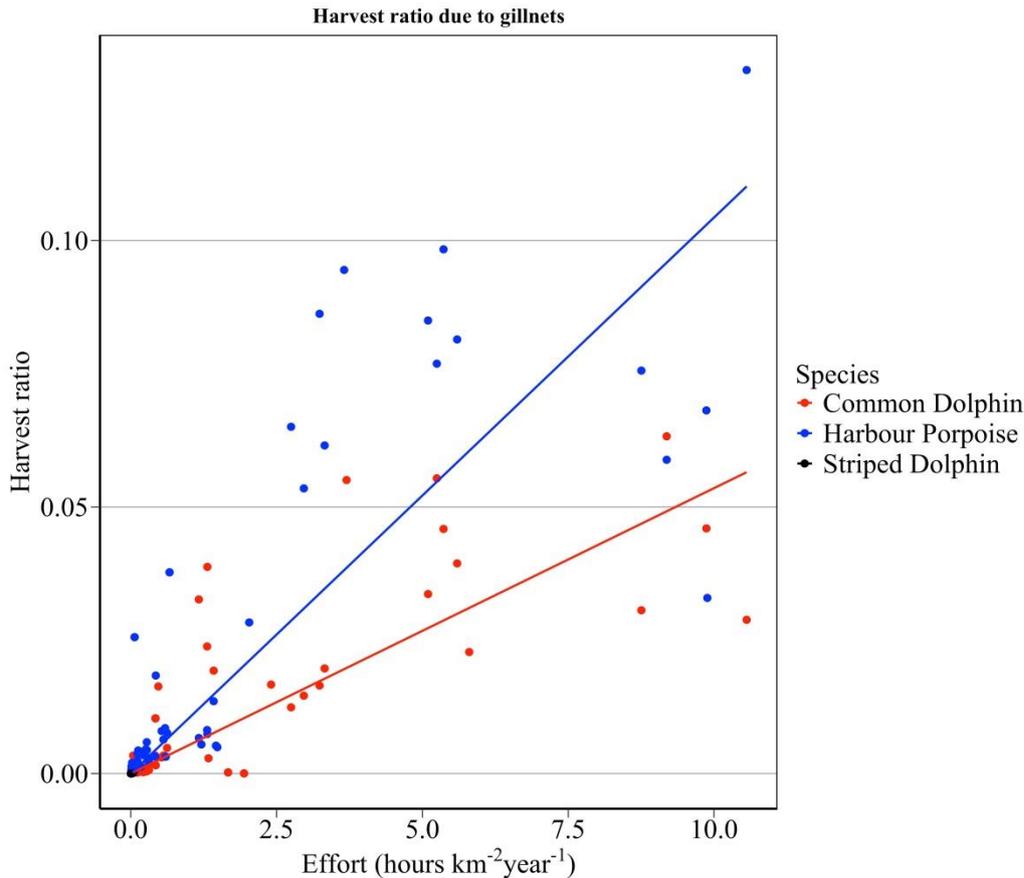


Figure A4.5. Scatterplots of the relationship between demersal gillnet effort and the harvest ratio (y^{-1}) on cetacean species. For each species, each point represents an individual national assessment of by-catch and effort in a given region.

Parameter optimisation for StrathE2E

The StrathE2E software package includes a parameter optimisation scheme which fits stationary states of the model to a wide range of observable properties of the ecosystem. The methodology involves likelihood estimation and a simulated annealing scheme using the Metropolis-Hastings iterative algorithm [4.6]. The target data to which the model is fitted are a set of indices of the real-world state of the ecosystem and their associated uncertainties (standard deviations). The indices may include annual average biomasses or concentrations, annual integrated production rates of food web components, or annual integrated dietary intakes. For each index, we need to be able to derive a corresponding value from the model output. For the North Sea model, the measures of ecosystem state and corresponding model driving data were based on two separate periods: 1970-1999, and 2003-2013. The model was required to fit to both of these periods with a common parameter set. The two periods were defined by their environmental conditions (temperature, ocean inflows etc.) and fishing fleet activity rates.

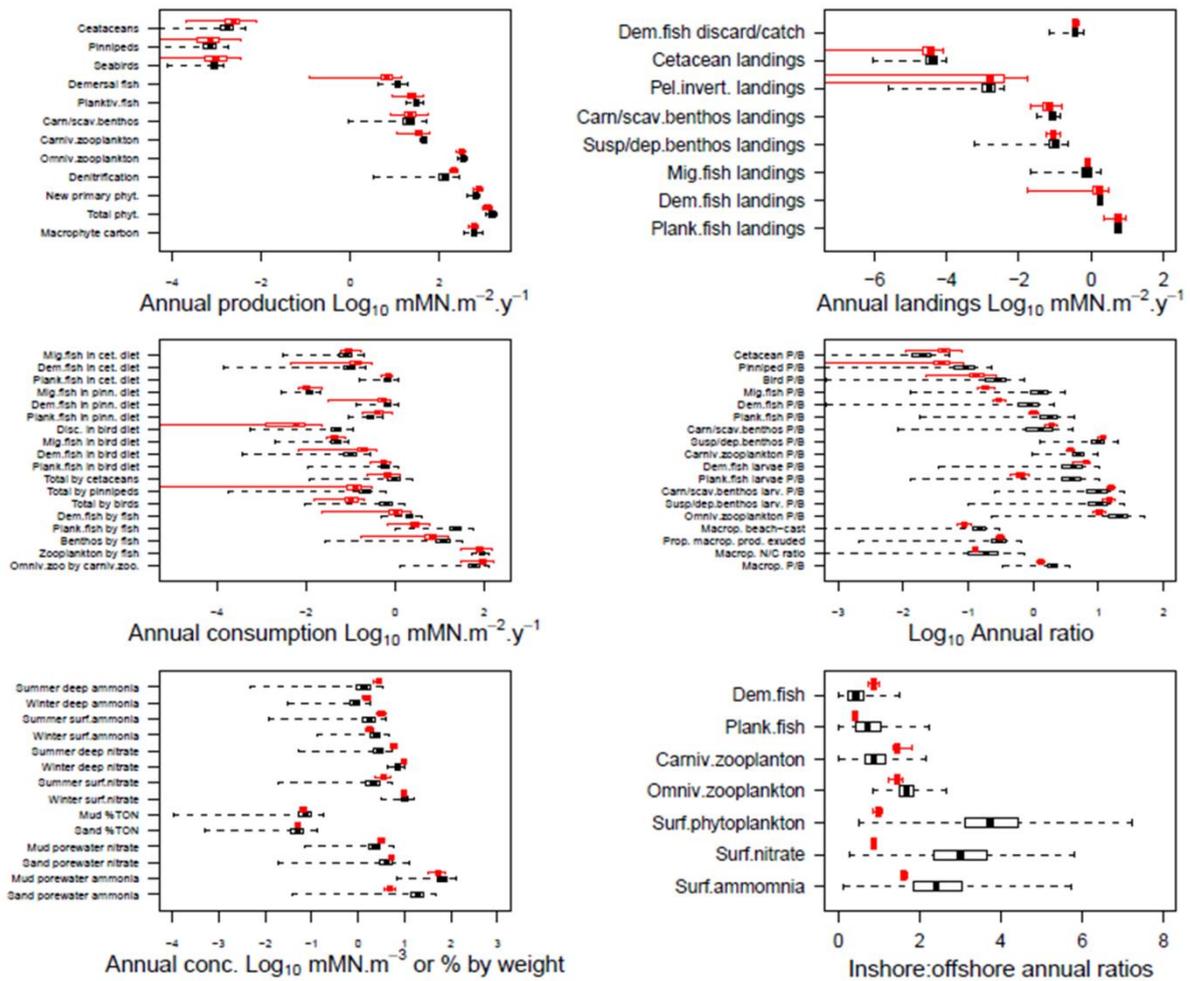


Figure A4.6. Best-fit model of the 1970-1999 North Sea ecosystem with the StrathE2E food web model. Each panel shows a different group of measures of the state of the ecosystem. The black box and whisker plots show the 0.5, 25, 50, 75 and 99.5 centiles of the observed data. The red plots show the equivalent statistics for the best-fit model results.

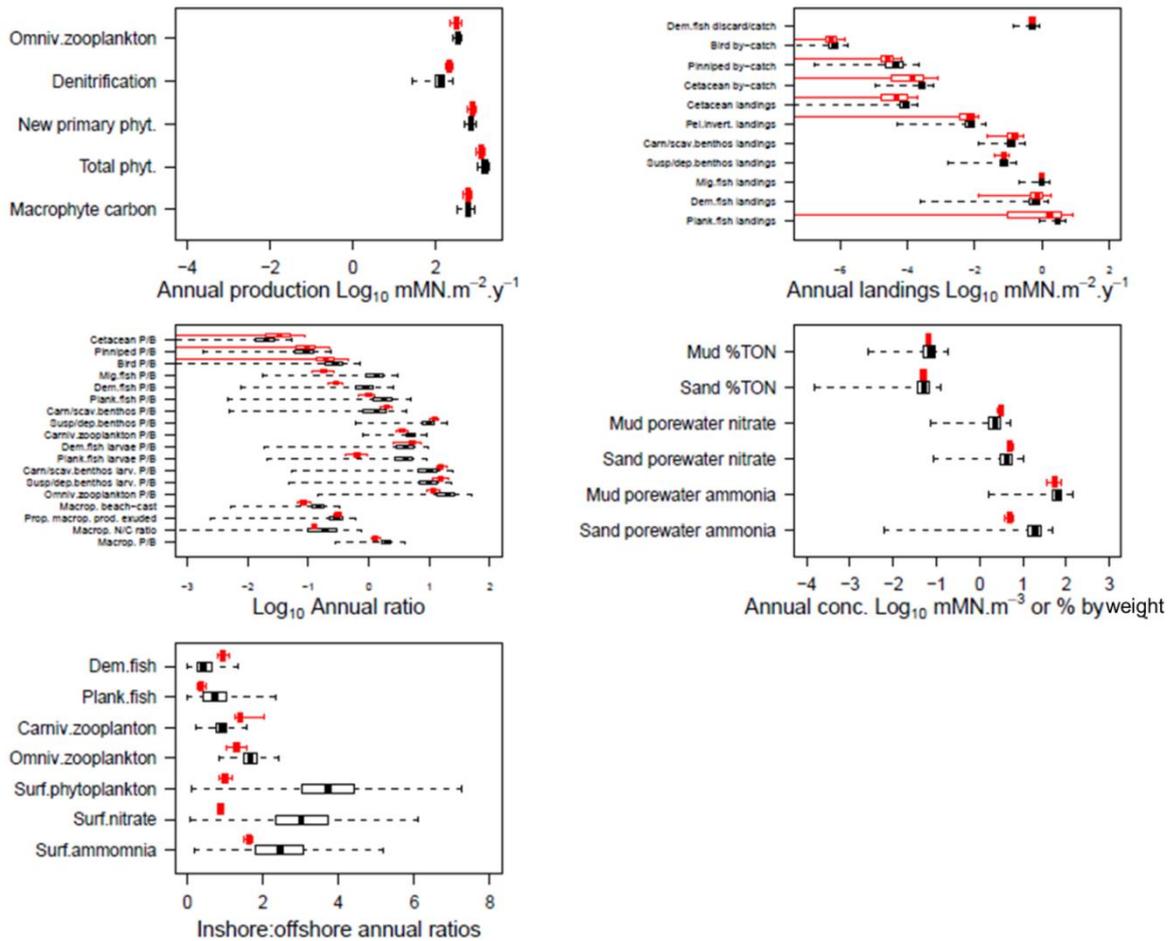


Figure A4.7. Best-fit model of the 2003-2013 North Sea ecosystem with the StrathE2E food web model. Each panel shows a different group of measures of the state of the ecosystem. The black box and whisker plots show the 0.5, 25, 50, 75 and 99.5 centiles of the observed data. The red plots show the equivalent statistics for the best-fit model results.

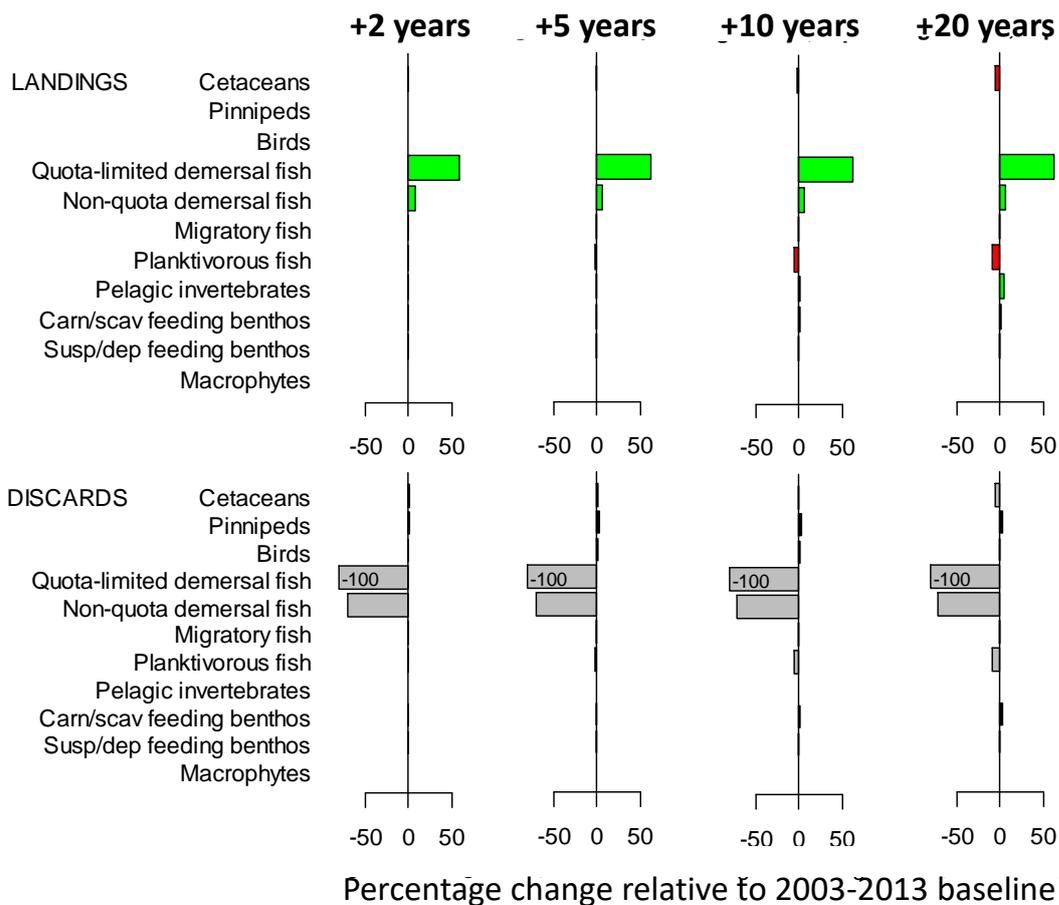


Figure A4.8. Simulated progression (years 2, 5, 10 and 20) of changes in the landings and discards from the North Sea foodweb following the full implementation of the Landing Obligation ('improved selectivity') (beginning of year 1) with status-quo fishing effort (successful outcome of negotiations) relative to a baseline state in 2003-2013. Each bar represents the percentage change in annual average landings (upper row, green or red bars) and discards (lower row, black or grey bars). Green bars indicate an increase in landings, red bars a decrease relative to the baseline. Black bars represent an increase in discards relative to the baseline, grey bars a decrease. For birds, seals and cetaceans, 'discards' refer to the incidental by-catch of these groups in fishing gears. The landings of cetaceans refer to catches of Minke whales taken by Norwegian whaling vessels.

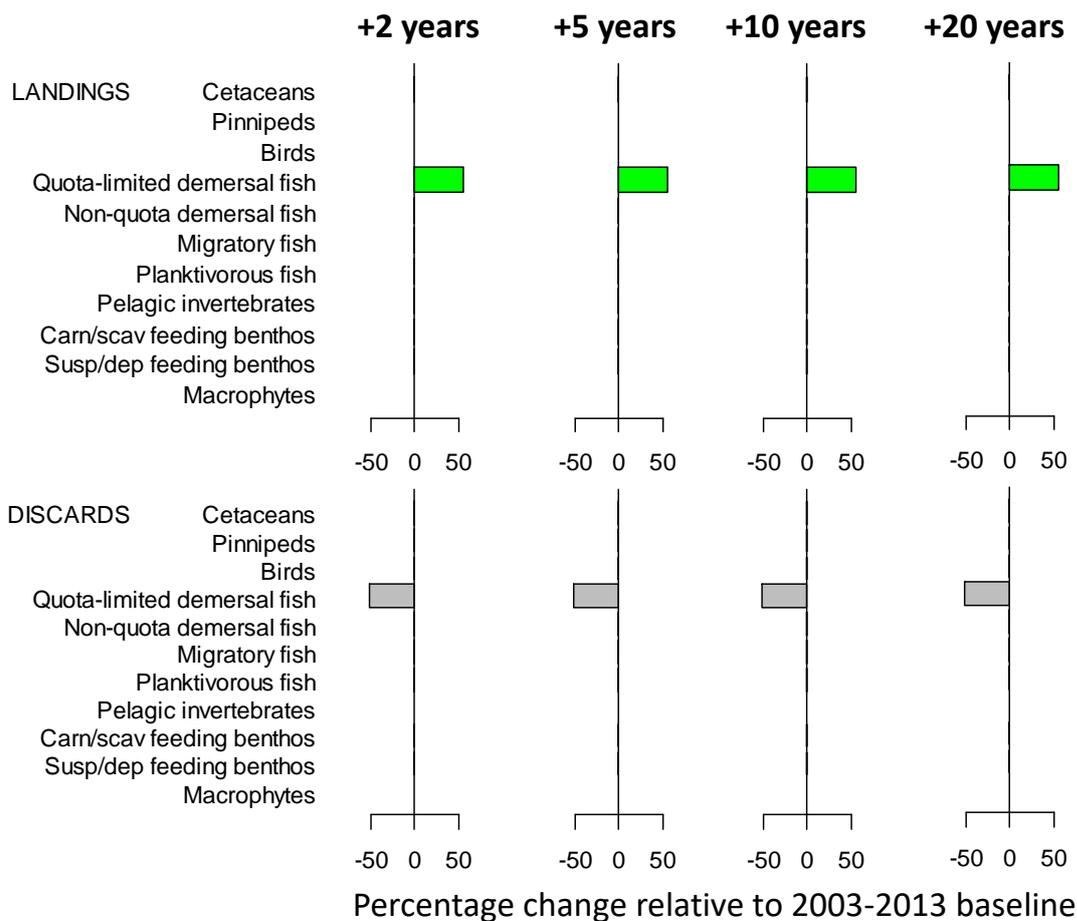


Figure A4.9. As Figure A4.8 with status-quo fishing effort (successful outcome of negotiations) relative to a baseline state in 2003-2013, but the ‘only-undersize’ discarding scenario.

References for Appendix 4

- [4.1] Heath, M.R. 2012. Ecosystem limits to food web fluxes and fisheries yields in the North Sea simulated with an end-to-end food web model. *Progress in Oceanography (Special issue: End-to-end modelling: Towards Comparative Analysis of Marine Ecosystem Organisation)* **102**, 42-66.
- [4.2] Heath, M., Wilson, R. & Speirs, D. 2015. FIS003 - Modelling the whole-ecosystem impacts of trawling. A study commissioned by Fisheries Innovation Scotland (FIS) <http://www.fiscot.org/> 86pp.
- [4.3] Beddington, J.R. 1975. Mutual interference between parasites or predators and its effect on searching efficiency. *Journal of Animal Ecology* **51**, 331-340.

[4.4] DeAngelis, D.L., Goldstein, R.A.. & O'Neill, R.V. 1975. A model for trophic interaction. *Ecology* **56**, 881-892.

[4.5] ICES 2018. Report from the Working Group on Bycatch of Protected Species (WGBYC), 1-4 May 2018, Reykjavic, Iceland. ICES CM 2018/ACOM:25. 128pp.

[4.6] Bertsimas, D. & Tsitsiklis, J. 1993. Simulated annealing. *Statistical Science* **8**, 10-15.

APPENDIX 5. R-scripts to generate the StrathE2E North Sea food web model results presented in this report.

```
# R code for application of the StrathE2E2 package in the project:
# 'Risks to fish stocks and wildlife in the North Sea posed by failure of post-Brexit fishery
# negotiations to reach agreement on quotas and access to UK waters".
#
# Requires R version 3.6 or later (https://www.r-project.org/)
# Requires the following R packages to be installed:
#   StrathE2E2      available from https://gitlab.com/MarineResourceModelling/StrathE2E (version 2.0.0)
#   deSolve        available from any CRAN server
#   NetIndices     available from any CRAN server
#
# To use this code, save it to a filename e.g. brexitmodels.R
# Then open an R session and type ... source("brexitmodels.R")
# Users will then have access to two functions for creating and viewing model results:
#   run.models()
#   compare.scenarios()
# For instructions on how to use these functions see the descriptions further down this file.
#
# Author: Michael Heath
#   University of Strathclyde, Department of Mathematics and Statistics
#-----
#
#
# The code sets up the StrathE2E model to represent the following scenarios:
#-----
# Scenario | Gear activity rates | Discarding conditions
#-----
# Baseline | 2003-2013 data | 2003-2013 data
# 1 | As in the baseline model (successful negotiation outcome) | 'Improved selectivity'
# 2 | As in the baseline model (successful negotiation outcome) | 'Only-undersize discarding'
# 3 | Negotiation-breakdown increments in gear activity rates added to the baseline | 'Improved selectivity'
# 4 | Negotiation-breakdown increments in gear activity rates added to the baseline | 'Only-undersize discarding'
#-----
#
#Discarding conditions are set in the model by switches in the model configuration:
```

```

# $fleet.model$DFsize_SWITCH      # = 0 in the default model
# $fleet.model$DFdiscard_SWITCH   # = 1 in the default model
# Details of the meaning of these settings is contained in the documentation accompanying the model package
#
# Switch setting in the fishing fleet parameters
#-----
# Scenario      | DFsize_SWITCH      | DFdiscard_SWITCH
#-----
# Baseline      | 0                  | 1
# 1             | 1                  | 0
# 2             | 0                  | 0
# 3             | 1                  | 0
# 4             | 0                  | 0
#-----
#
# The scenarios representing breakdown of post-Brexit fisheries negotiations involve applying multipliers to the activity rates
# of the various fishing gears in the model. These reflect the changes in total international activity that would arise
# as a result of such a breakdown:
#
# Gear multipliers to generate negotiation breakdown conditions
actmult_brexit <- c(1.581, # Pelagic_Trawl+Seine
                  1.059, # Otter30-70mm+TR3(sandeel+sprat)
                  1.202, # Longline_mackerel
                  1.103, # Beam_Trawl_BT1+BT2
                  1.048, # Demersal_Seine
                  1.1,   # Demersal_Otter_Trawl_TR1
                  1.07,  # Gill_Nets+Longline_demersal
                  1.004, # Beam_Trawl_shrimp
                  1.001, # Nephrops_Trawl_TR3
                  1,     # Creels
                  1.007, # Mollusc_Dredge
                  1)    # Whaler
#-----

library(StrathE2E2) # Load the StrathE2E2 package

```

```

#Set up the model configurations
#-----

#Baseline model
basemodel<-read_model("North_Sea","2003-2013",model.ident="baseline")

#Scenario 1 - successful negotiations, improved selectivity
scenario1model<-basemodel
scenario1model$data$fleet.model$DFsize_SWITCH    <- 1      # changed from 0 in the baseline model
scenario1model$data$fleet.model$DFdiscard_SWITCH <- 0      # changed from 1 in the baseline model
scenario1model$setup$model.ident                 <- "brexitscenario_1"

#Scenario 2 - successful negotiations, undersize catch discarded
scenario2model<-basemodel
scenario2model$data$fleet.model$DFsize_SWITCH    <- 0      # unchanged from 0 in the baseline model
scenario2model$data$fleet.model$DFdiscard_SWITCH <- 0      # changed from 1 in the baseline model
scenario2model$setup$model.ident                 <- "brexitscenario_2"

#Scenario 3 - breakdown of negotiations, improved selectivity
scenario3model<-basemodel
scenario3model$data$fleet.model$gear_mult <- actmult_brexit
scenario3model$data$fleet.model$DFsize_SWITCH    <- 1      # changed from 0 in the baseline model
scenario3model$data$fleet.model$DFdiscard_SWITCH <- 0      # changed from 1 in the baseline model
scenario3model$setup$model.ident                 <- "brexitscenario_3"

#Scenario 4 - breakdown of negotiations, undersize catch discarded
scenario4model<-basemodel
scenario4model$data$fleet.model$gear_mult <- actmult_brexit
scenario4model$data$fleet.model$DFsize_SWITCH    <- 0      # unchanged from 0 in the baseline model
scenario4model$data$fleet.model$DFdiscard_SWITCH <- 0      # changed from 1 in the baseline model
scenario4model$setup$model.ident                 <- "brexitscenario_4"

#-----

# ~~~~~

```

```

# ~~~~~
# ~~~~~

# FUNCTION run.models()
#-----
run.models<-function(yy=20){
# Description:
#     Runs the baseline and each of the 4 scenario models for a selected number of years
#     The function can take up to a few minutes to complete depending on the number of years to run
#
# Arguments:
#     yy     Number of year to run each of the 5 models (default=20)
#
# Returns:
#     List object comprising the run results from the baseline and each of the 4 scenario models
#
# Example:
#     rundata<-run.models()           # runs each of the models for 20 years
#
#     rundata<-run.models(yy=10)     # runs each of the models for 10 years

baseresults<-StrathE2E(basemodel,nyears=yy, csv.output=FALSE)
cat("Completed baseline model run for ",yy," years", "\n")
scenario1results<-StrathE2E(scenario1model,nyears=yy, csv.output=FALSE)
cat("Completed scenario 1 model run for ",yy," years", "\n")
scenario2results<-StrathE2E(scenario2model,nyears=yy, csv.output=FALSE)
cat("Completed scenario 2 model run for ",yy," years", "\n")
scenario3results<-StrathE2E(scenario3model,nyears=yy, csv.output=FALSE)
cat("Completed scenario 3 model run for ",yy," years", "\n")
scenario4results<-StrathE2E(scenario4model,nyears=yy, csv.output=FALSE)
cat("Completed scenario 4 model run for ",yy," years", "\n")

run_results<-list(baseresults=baseresults,
                  scenario1results=scenario1results,
                  scenario2results=scenario2results,
                  scenario3results=scenario3results,

```

```

        scenario4results=scenario4results)

# End of function run.models
}

# ~~~~~
# ~~~~~
# ~~~~~

# FUNCTION compare.scenarios()
#-----
compare.scenarios <-function(inputdata,scenarioA,scenarioB,type){
# Description:
#     Creates a tornado plot comparing two of the scenario runs
#
# Arguments:
#     inputdata      Name of a list object created by the run.models function
#     scenarioA      ID of the scenario to be treated as a baseline (values 0, 1, 2, 3 or 4; 0 = baseline model)
#     scenarioB      ID of the scenario to be treated as a response (values 0, 1, 2, 3 or 4; 0 = baseline model)
#     type           Text strings "aam" or "catch" to select annual average mass or catch comparison
#
# Returns:
#     List object comprising the numeric data corresponding to the selected plot.
#     See "help(compare_two_runs_aam)" from an R command prompt for more details
#
# Examples:
#     scen31acomp<-compare.scenarios(rundata,1,3,"aam") # Compares annual average mass results for
#                                                       scenario 3 in the run results object 'rundata'
#                                                       with the results from scenario 1
#     scen31acomp # View the numeric data for the comparison
#
#     scen31ccomp<-compare.scenarios(rundata,1,3,"catch") # Compares annual catch results for
#                                                         scenario 3 in the run results object 'rundata'
#

```

```

#                                     with the results from scenario 1
#   scen31ccomp                       # View the numeric data for the comparison
#
# To open a new graphics window for the screen output...
#   dev.new()
#   scen10acomp<-compare.scenarios(rundata,0,1,"aam") # As above but for sceanrio 1 vs the baseline model
#
# To direct the graphics output to a file in the current user workspace rather than to the screen...
#   pdf("plot.pdf",width=6,height=8) # or jpeg("plot.jpg",...), png("plot.png",...)
#   scen10acomp<-compare.scenarios(rundata,0,1,"aam")
#   dev.off()

#Check the function arguments for valid values and halt the function is invalid
validates<-0
validatet<-0
if(scenarioA==0 | scenarioA==1 | scenarioA==2 |scenarioA==3 |scenarioA==4) validates<-1
if(scenarioB==0 | scenarioB==1 | scenarioB==2 |scenarioB==3 |scenarioB==4) validates<-1
if(type=="aam" | type=="catch") validatet<-1
if(validates==0){
  stop("Arguments 'scenarioA and scenarioB' must have numeric values 0, 1, 2, 3 or 4")
}
if(validatet==0){
  stop("Argument 'type' must be text strings aam or catch (in inverted commas)")
}

#Extract the baseline and scenario results from the inputdata list
if(scenarioA==0){
  scenarioAresults<-inputdata$baseresults
  titleA<-"2003-2013 baseline"
}
if(scenarioA==1){
  scenarioAresults<-inputdata$scenario1results
  titleA<-"Successful negotiation"
}
if(scenarioA==2){
  scenarioAresults<-inputdata$scenario2results
  titleA<-"Successful negotiation"
}

```

```

}
if(scenarioA==3){
  scenarioAresults<-inputdata$scenario3results
  titleA<-"Negotiation breakdown"
}
if(scenarioA==4){
  scenarioAresults<-inputdata$scenario4results
  titleA<-"Negotiation breakdown"
}

if(scenarioB==0){
  scenarioBresults<-inputdata$baseresults
  titleB<-"2003-2013 baseline"
}
if(scenarioB==1){
  scenarioBresults<-inputdata$scenario1results
  titleB<-"Successful negotiation, improved selectivity"
}
if(scenarioB==2){
  scenarioBresults<-inputdata$scenario2results
  titleB<-"Successful negotiation, undersize discards"
}
if(scenarioB==3){
  scenarioBresults<-inputdata$scenario3results
  titleB<-"Negotiation breakdown, improved selectivity"
}
if(scenarioB==4){
  scenarioBresults<-inputdata$scenario4results
  titleB<-"Negotiation breakdown, undersize discards"
}

if(type=="aam"){
scencomp <- compare_two_runs_aam(modell1=NA,results1= scenarioAresults,from.csv1=FALSE,
                                model2=NA,results2=scenarioBresults,from.csv2=FALSE,
                                log.pc="PC", zone="W",
                                bpmin=(-40),bpmax=(+40),
                                maintitle=paste(titleB, " vs ",titleA))

```

```
}
if(type=="catch"){
scencomp <- compare_two_runs_catch(model1=NA,results1= scenarioAresults,from.csv1=FALSE,
                                model2=NA,results2=scenarioBresults,from.csv2=FALSE,
                                log.pc="PC", zone="W",
                                bpmmin=(-80),bpmmax=(+80),
                                maintitle= paste(titleB," vs ",titleA))
}

scencomp

# End of function compare.scenarios

}

# ~~~~~
# ~~~~~
# ~~~~~
```