The role of discarding in the dynamics of the demersal fish community of the Firth of Clyde

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Abstract

The Firth of Clyde is one of the main grounds of the Scottish Nephrops (*Nephrops norvegicus*, or Norway lobster) trawl fishery. The fishery takes a bycatch of demersal fish that includes cod, haddock and whiting. Almost 100% of these fish are discarded due to trawlers not possessing licenses to land whitefish or because the fish are below the minimum conservation reference size. Even though targeted fishing for whitefish had effectively ceased by the early 2000s, there are still no signs of stock recovery in the Clyde.

One hypothesis for the lack of recovery is that fish discards in the Nephrops trawl fishery are sufficient to maintain a high mortality rate on the stocks, despite the best efforts from the industry to minimize bycatch. To explore this hypothesis, quantities of cod, haddock and whiting discarded by the Nephrops fishery in the Clyde were estimated. Second, an age-structured stock assessment model was developed that uses scientific survey data and commercial fisheries data to estimate fishing mortality rates and stock biomass, and applied to the three main species of whitefish in the Clyde. Lastly, the populations were projected forward 30 years, under different mortality and recruitment scenarios to identify potential stock recovery.

Results from the assessment model show historically high levels of mean fishing mortality for all three stocks and low levels of spawning biomass. Projections suggest that only haddock has some chance of recovery under current fishing conditions. For whiting and cod stocks, the projections show recovery only when substantial reductions in fishing mortality are made where the strength of recovery is dependent on the level of recruitment. Auto-correlated discard estimates were used as model inputs which may lead to stable but unrealistically precise model fits in the more recent years (after 2002). Further work on this more recent period that explores alternative discard estimates would be useful to characterise uncertainties in stock recovery. Prior to 2002, when the age-based assessment uses only survey data, fishing mortality rates are also estimated to be high and stock biomass declines. This is consistent with fishing mortality from bycatch in the Nephrops fishery being a likely significant factor in the lack of recovery of the cod and whiting stocks in the Clyde.

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CHAPTER 1 - Introduction

1.1 Overview - Clyde Marine Region

The Firth of Clyde is a large semi-enclosed basin at the Scottish west coast. It can be divided in two areas with distinct physical characteristics – an inner Firth composed of narrow and long sea lochs, also containing the Clyde estuary, and a more exposed outer Firth of open waters around the isle of Arran (Figure 2.1 in Chapter 2). The Clyde has been subject to human exploitation pressures for many centuries due to its productivity and proximity to coastal settlements.

Several economic, social and conservation interests lead to the establishment of a regional group in 2016 to prepare a management plan for the Clyde Marine Region. The Clyde Marine Planning Partnership (CMPP) is composed of government bodies, fishing industry, marine transport and aquaculture sectors, nature conservation organizations, and advised by a scientific committee of academics from Scottish universities. It has authority in all matters related with marine planning in the Clyde, and involves both the stakeholders and public input through public consultations. One of the first goals of the CMPP was the "Clyde Marine Region Assessment 2017" (Mills et al., 2017). This extensive report details physical and environmental characteristics of the region (e.g. seabed sediments, water circulation and oceanographic properties, climate regime, marine fauna and flora) and identifies and classifies significant pressures of human activities, mainly shipping and transport, recreation and tourism, contamination and pollution sources, aquaculture and fishing. It also brings attention to the historic and cultural heritage of sea and land-based activities, and identified knowledge gaps and research needs.

The work of the CMPP built on a large body of existent work, including the "Clyde Ecosystem Review" from Marine Scotland Science (McIntyre et al., 2012) and the "State of the Clyde – Environment Baseline Report" from SSMEI (Scottish Sustainable Marine Environment Initiative, Clyde Pilot; Ross et al., 2009). The CMPP report lists the full range of the Clyde's Marine Protected Area (MPA) network. The report prepared by the Sustainable Inshore Fisheries Trust (SIFT, 2015) provides an extensive review of the spatial management measures currently in place in the Firth of Clyde. Three MPAs were established in 2014 in locations where sensitive habitats and species are present. These are in Upper Loch Fyne, designated for protection of flame shell beds and horse mussel beds; the Clyde Sea Sill, on the shallow waters at the entrance of the Clyde region that contain islands with seabird colonies; and South of Arran, that contains maerl beds, seaweed and kelp communities, and seagrass beds (map 18 in McIntyre et al., 2012). These habitats and associated benthic fauna are particularly vulnerable to sea bottom disturbance from towed fishing gears, and can take up to 15 years until showing any signs of recovery after trawling or dredging activities (Collie et al., 2000). There are different restrictions in separate sections of the MPAs; trawling and dredging are prohibited in sections of South Arran and Loch Fyne MPAs but allowed (with restrictions) in other areas (map 19 in Mills et al 2017).

The impact of trawling activities in the Clyde has been linked to resuspension of sediments, reduction of overall habitat and community diversity, and an increase in habitat similarity between different locations (Coggan et al., 2001). MPAs and closed areas show potential for benthic community and fish stock recovery. In the Clyde, closed areas such as the no take zone in South of Arran (Lamlash bay) was shown to increase recruitment and the number of individuals of commercially important species such as scallops when compared with areas outside the closed area (Howarth et al., 2015). More than 80% of the Clyde seabed is composed of varying proportions of muddy and sandy sediments, making it an ideal ground for the burrowing crustacean Norway lobster (*Nephrops norvegicus*). The Nephrops fishery began in early 1950's and is now the most important fishery economically. The next sections give an overview of the history of fisheries in the Clyde and changes in the demersal fish community over two centuries.

1.2 Fisheries in the Clyde

1.2.1 History of fisheries in the Clyde The fishing industry has a major historical, economic and cultural importance for the Clyde rural communities. In the past three centuries (for which there are records), there had been various fisheries for pelagic, demersal fish and shellfish. The herring (*Clupea harengus*) fishery in the Clyde was the traditional mainstay in the 19th and mid-20th centuries, with fishing grounds concentrated in the inner Clyde for autumn-spawned herring and in Ballantrae Bank for spring-spawned herring (Bailey et al., 1986a). The fishery used trammel and drift nets but gradually changed to seine or ring nets by mid-1850's, not without a dispute between fishers that used each gear type. Average herring landings in the period 1893 – 1960 varied around 14 200 tonnes per year. At the same time, a demersal fishery for gadoids and flatfish operated with handlines and longlines, mostly during the summer months. Historical accounts compiled by Jones et al. (2015) described a decline of whitefish landings in the Clyde during the second half of the 19th century. By the end of the century, steam powered trawlers came into regular use, and were accused of depleting inshore fish populations, which lead to a trawling ban within the Clyde from 1889 onwards.

Further technological improvements allowed the herring fishery to thrive during the first half of the 20th century. However, by mid-century, the fishery was dependent on immature herring and shoals were observed less frequently. The pelagic pair trawl (known as "light trawl") was introduced in 1960's. The light trawl was more efficient in catching less dense schools of herring, and allowed diversification of catches into other stocks such as demersal fish species, scallops and Nephrops (Thurstan and Roberts, 2010). The trawling ban was partially lifted in 1962 for smaller vessels, allowing demersal trawlers to operate outside 3 nautical miles off the coastline within the Firth of Clyde. By 1971, annual landings of herring were less than a quarter of the long-term average, due to a lack of recruitment to the local spring-spawning stock. The herring population did not recover, despite the controls that were imposed at the time (seasonal closures and implementation of total allowable catches, or TACs). The seine net herring fishery was replaced by demersal trawling for gadoids and Nephrops. The fishermen deployed either a light trawl or a prawn trawl, depending on the net mesh size and the proportions of whitefish/Nephrops in the catch.

Between 1970 and 1980, five species of demersal fish accounted for 80% of total weight of landings in the Clyde. These were cod (*Gadus morhua*), whiting (*Merlangius merlangus*), saithe (*Pollachius virens*), haddock (*Melanogrammus aeglefinus*) and hake (*Merluccius merluccius*). Other regularly landed species in minor quantities were spurdog (*Squalus acanthias*), plaice (*Pleuronectes platessa*), skates and rays (Rajidae), and monkfish (*Lophius piscatorius*). These main five whitefish stocks were exploited intensively, and the fishery depended largely on younger age groups, making it sensitive to variations in the strength of recruiting year classes (Hislop, 1986). By early 1980's the landings of the main commercial fish species had declined substantially (Thurstan and Roberts, 2010).

In 1984, the 3 nautical mile limit was removed in an effort to sustain fish landings, but these continued to decline. At the same time, Nephrops catches were increasing, and a surge in Nephrops abundance occurred probably due to reduced abundance of predators such as cod and haddock. By the 21st century, demersal fish landings had declined to a negligible part of profits from the Clyde fisheries, as most fishing effort is concentrated on Nephrops and other shellfish species.

1.2.2 The Nephrops fishery in the Clyde The Nephrops fishery in the Clyde started in the 1950's with a modified seine net that was used previously for catching herring (Bailey et al., 1986b), since trawling was forbidden before 1960's. Over time, with trawling restrictions being lifted, the fishery adopted a more typical funnel shaped trawl net with otter boards to keep the net open, and a bottom rope with heavy rubber discs to drag it along the seafloor. The Nephrops trawling fishery rapidly expanded to the most important fishery in revenue terms. In 1984, it landed 3000 tonnes of Nephrops worth £4 million, representing over 20% of the value of Scottish Nephrops landings as a whole (Bailey et al., 1986b). Values from 2014 show 5700 tonnes of Nephrops landed, with a value of £14.8 million and accounting for 24% of total Nephrops landed in Scotland (McIntyre et al., 2012; Russell and Mardle, 2017). The most important landing ports are in Campbeltown and Tarbert in the Kintyre peninsula, and Troon and Saltcoats in Ayrshire.

There are currently two fleet segments catching Nephrops in the Firth of Clyde, operating with distinct fishing gears. Besides the trawlers, a fishery with creels started to develop during 1980's in areas where trawling was either impossible or prohibited. The creels are static fishing gear, which are baited and deployed on the sea bed, and retrieved after 1-2 days to extract the catch. The trawlers are active/mobile fishing gears, and can consist of single rig or twin rig trawl nets (two parallel nets deployed side by side), depending on the size of the vessel. Most of the commercial activity in the Clyde is undertaken by vessels under 15 metres (McIntyre et al., 2012). Including boats from Northern Ireland, approximately 164 vessels participated in the Clyde Nephrops fishery as of 2012, both trawlers and creel vessels (Ryan and Bailey, 2012).

Nephrops landed in Scotland are sold on the domestic and overseas markets, either in whole form or as tails (Russell and Mardle, 2017). Creel-caught Nephrops are mostly sold whole and alive in tubes, attaining higher prices (two times higher) than trawl-caught Nephrops. Trawl-caught Nephrops are a mix between whole and tails (tailed Nephrops are processed at sea by removing the cephalotorax), and often sold frozen.

There is an ongoing conflict for space on fishing grounds between fishers of creel vessels and trawlers in Scottish Nephrops grounds. A weekend ban on mobile gear helped to alleviate this issue, but competition still exists, even between fishers that employ the same fishing gear. Until 1999, landings had to constitute at least 30% by weight of Nephrops when using single-rig trawls, but nowadays the fishery has shifted to targeting Nephrops only, so this regulation is likely to be outdated (Stratoudakis et al., 2001). Other technical measures in place are minimum landing sizes for Nephrops (total length of 70 mm, carapace length of 20 mm and tail size at 37 mm), and the introduction of a square mesh panel at the top of the trawl net to allow juvenile fish to escape. The mesh sizes increased from 70 to 80 mm in 2009, with an increase of 80 to 120 mm for the square mesh panel (Ryan and Bailey, 2012). These changes were part of a wider programme instated by the European Union's "Cod Recovery Plan" to promote recovery of the cod stocks in European waters, and the adoption of the Scottish Conservation Credits scheme which controlled fishing effort of vessels catching cod in Scottish waters.

Previous studies on bycatch and discard composition on the Clyde Nephrops fishery estimated high rates of discarding (Stratoudakis et al., 2001; Bergmann et al., 2002a). Up to 70% of total catch was discarded and mainly composed of juvenile demersal fish like haddock and whiting. In the context of this work, discards are defined as bycatch organisms that are assumed to be returned to sea as they are considered undesirable, for economic or regulatory reasons.

Other commonly discarded species were poor cod (*Trisopterus minutus*), long rough dab (*Hippoglossoides platessoides*), hake and Norway pout (*Trisopterus esmarkii*; Stratoudakis et al., 2001). Bergmann et al. (2002a) found that trawls conducted in the south of the Clyde region generated larger quantities of whitefish bycatch, including cod, haddock and whiting, than trawls in the north of the region which contained higher proportions of invertebrates. This is likely to be a reflection of differing bathymetries, benthic communities and ground types. A large proportion of dead discards ends up in the seabed and becomes available as food source for benthic scavengers. Several epibenthic species utilise discards from the Clyde trawlers, with Nephrops being one of the most abundant megafaunal scavenger (Bergmann et al., 2002b). Seabird and marine mammal species are also associated with fishing boats, and discards and offal can represent a significant proportion of their diet (McIntyre et al., 2012; Heath et al., 2014).

The full implementation of the landing obligation in 2019 in Scottish fisheries attempted to put an end to discarding practices. Under the landing obligation, all catches of regulated species have to be kept on board, landed and counted against quotas. Landed undersized fish (fish below MCRS, minimum conservation reference size) cannot be sold for direct human consumption but can be used as bait, fish meal and other industrial purposes. An exemption on discarded Nephrops is applied in the Clyde creel fishery since it has high survival after being returned to sea.

Another type of exemption is the so called “de minimis” exemption, that allows for a small percentage of the total annual catches of a certain species to be discarded, where either improvement on gear selectivity is considered very difficult or when there are disproportionate costs of handling unwanted catches. The Nephrops trawl fishery in the Clyde has been allowed a “de minimis” exemption and may discard up to 7% of undersized Nephrops (Mills et al., 2017). For the West of Scotland bottom trawl fisheries, there are “de minimis” exemptions for horse mackerel and mackerel, greater silver smelt and haddock.

A fishery closure in the south of the Firth of Clyde was introduced in 2001 (Scottish Statutory Instrument, SSI) directed at protecting spawning aggregations of cod. The closure was proven to be effective in avoiding an increase in local fishing mortality as a result of fishing effort being displaced from a similar closure in the Irish Sea (Armstrong et al., 2005). However, it has not demonstrated any local recovery of cod biomass more than a decade after its implementation (Clarke et al., 2015). There is evidence that the Clyde demersal fish community and the whole ecosystem has changed over the past century at least. The next section gives a detailed account on what changes occurred and plausible causes.

1.2.3 Changes in the Clyde demersal fish community The Clyde has been described as an "ecosystem nearing the endpoint of overfishing, a time when no species remain that are capable of sustaining commercial catches". This work by Thurstan and Roberts (2010) attracted considerable media attention and blamed the 3-mile limit removal for the collapse of the demersal fish stocks up until this day. However, their analysis used landings data alone, which are not enough to understand what really happened in the Clyde demersal community. The landings do not contain the discarded portion of the catches, and are subject to other bias such as changes in gear technology, moving of fishing grounds, market demands, and decisions from the skippers of the vessels. McIntyre et al. (2012) raised a point that the decrease in fish landings in the Clyde can also be related with changes in annual average temperatures from climate change, but this relationship does not necessarily imply causation.

Using survey data which is collected by research vessels provides a complementary approach to the traditional use of commercial catch records. The survey data is collected with a GOV ("Grand Overture Vertical") net. The GOV net might not be as efficient as other demersal trawls, but the aim is to collect a representative sample of all the fish present in an area using a standardized method that is not altered from one year to the next. This way the survey data can be used as a time series. This is what was done by Heath and Speirs (2012). In this paper, the authors analysed changes in biomass density, species diversity and length structure of the demersal fish community in the Clyde between 1927 and 2009 from scientific survey data. The authors concluded that the Clyde remains a productive ecosystem, where the biomass of the main commercial species in late 2000's was double the biomass before 1960 when trawling started. However, the demersal fish community has undergone a transformation. The size distributions are truncated, with few marketable individuals after the period of peak landings in 1980's. In 1960's, the biomass was distributed among numerous species with top predators such as spurdog, thornback ray and cod, whereas in 1980-1990 only a few species dominated the biomass and were mostly whiting and haddock. The evenness indicator, which gives a measure of the species abundance across taxa, show a considerable decrease from 1980 to 2000. This evenness indicator has recovered in early 2000, meaning that species abundance was distributed over a wider range of taxa, but the species for which abundance increased are mostly small-growing taxa like Norway pout and poor cod. Lawrence and Fernandes (2021) reported a recent recovery of the pelagic biomass in the Clyde, now dominated by sprat (*Sprattus sprattus*) instead of herring. This shift from larger animals to smaller ones during the 20th century seems also to have occurred in the Irish Sea and in English coastal regions (Rogers and Ellis, 2000). Heath and Speirs (2012) compared the Clyde region with adjacent areas around the southern Hebrides and northern Irish Sea, and these changes on demersal community indicators were more accentuated in the Clyde. Changes in the Clyde Sea ecosystem are comparable to those that have occurred in marine ecosystems over the world. In the Black Sea, a combination of overfishing and eutrophication lead to multiple regime shifts. First, the decline of large predators and an increase of small pelagic fish stocks. Then, the collapse of small pelagic fish was followed by a rise of gelatinous zooplankton (Daskalov et al., 2007). The well-known collapse of the cod stocks on the Grand Banks off Newfoundland was due to overfishing in these eastern Canadian waters. Despite a moratorium on fishing, the cod stocks have not recovered and the fishery was closed in 2003. The fishing effort was mostly redirected to profitable crustacean fisheries for species like northern shrimp and American lobster, a similar avenue of fisheries history as in the Clyde (Hamilton et al., 2004).

There are multiple reasons that potentially contributed to the transformation of the Clyde whitefish community. Hunter et al. (2015) investigated changes in typical lengths at maturation of cod, haddock and whiting in the Firth of Clyde, and compared them with the wider west coast. The lengths at maturation of haddock, whiting and female cod decreased significantly between 1986 and 2009, with rates of change being particularly accentuated in the Clyde as compared to adjacent areas. These changes were partially attributed to fishing, since fishing intensity in the Clyde is much greater than in adjacent areas. Hunter et al. (2016) show that growth rates and maximum lengths of haddock and whiting in the Clyde have decreased significantly from 1980 to 2012, and at a faster rate than in the wider Scottish west coast. Both these works show that whitefish in the Clyde are more likely to mature at smaller lengths and reach smaller maximum lengths, which could be one reason for explaining the current state of the demersal fish community.

Another hypothesis is related with habitat associations of gadoids during their juvenile stages. In the South Arran MPA, stereo-video observations found that cod was most abundant in shallow, sheltered areas composed of gravel-pebble substrates that contained maerl (Elliott et al., 2016; Elliott et al., 2017a; Elliott et al., 2017b). In contrast, haddock and whiting occurred in higher abundances in more homogeneous habitats composed of mud and sandy substrates. Much of the maerl beds found were degraded as a result of historical dredging activities. This link between epibenthos and demersal gadoids has significant implications if insufficient suitable habitats may be causing impaired recruitment, particularly for cod in the Clyde. Furthermore, bait experiments showed that haddock and whiting juveniles were most attracted to bait, displaying a scavenging behaviour, while cod juveniles were not attracted to bait (Elliott et al., 2018). The authors reasoned that this might translate into better survival strategies for haddock and whiting during juvenile stages. There are other environmental factors that could contribute to the changes of the demersal fish community in the Clyde. Excess nutrients and pollutants from industry and urbanisations, shipping activities, and discharges of dredged material all have an impact on the water quality of the region (Mills et al., 2017; McIntyre et al., 2012). Climate change and warming seas will certainly have an impact on fish populations, with effects already being observed in areas like the North sea of shifting species distributions, smaller body sizes and faster life cycles (Perry et al., 2005; Baudron et al., 2014; Ikpewe et al., 2020), and affecting stock-recruitment relationships (Cook and Heath, 2005).

One more directly quantifiable proposition for the lack of recovery of the whitefish community after 20 years of non-targeted fishing can be that the bycatch from the Nephrops fishery is suppressing the recovery of these stocks. The Nephrops fishery in the Clyde continues to catch demersal gadoids together with its target species, and at times in large quantities. Despite best efforts from the industry to reduce bycatch through increases in mesh sizes and introduction of square mesh panels, this bycatch might still represent a high, but hidden, mortality rate on the stocks. An ecosystem modelling approach for the whole of the West Coast has shown that there is insufficient bycatch from the Nephrops fleet to have a large impact on gadoid stocks (Alexander et al., 2015). However, the analysis was conducted at the level of the west coast, and it has been shown that the Clyde may respond distinctively than neighbouring areas (Heath and Speirs, 2012; Hunter et al., 2015; Hunter et al., 2016). In fact, other works have suggested that fish bycatch by Nephrops trawlers might be having a negative impact on demersal stocks in the Clyde (McIntyre et al., 2012; Clarke et al., 2015; Thurstan and Roberts, 2010; Burns et al., 2019).

This thesis sets out to investigate the hypothesis that fish bycatch in Nephrops trawlers in the Clyde is suppressing the recovery of the whitefish populations, mainly of cod, haddock and whiting. To answer this question, fish population models are developed and applied to the Clyde to estimate fishing mortality and fish biomass. In order to develop the analytical models, the theoretical background on population dynamics in the Clyde and adjacent areas are reviewed in the following section. It starts with a brief overview of stock assessment methodology, the current situation of stock assessment and advice on the West coast of Scotland, and finishes with aspects and literature on stock unit identification and connectivity between Clyde and adjacent stocks.

1.3 Fish population dynamics in the Clyde and adjacent stocks

1.3.1 Stock assessment methods - brief review Fisheries management and the sustainable exploitation of fish stocks depend on the evaluation of the state of fish stocks. Quantities such as spawning stock biomass and fishing mortality are estimated using available data from fisheries-dependent and independent sources and compared to pre-agreed reference points to evaluate stock status. A stock assessment can be used to forecast stock dynamics and predict catches that satisfy management goals and predicting the consequences of future management scenarios. These assessments are based on mathematical and statistical models. There are various steps in the stock assessment process: data collection and processing; preparation of input data, model selection, estimation and forecasting; communication of results and recommendations to fisheries managers.

Stock assessment methods started to be developed in early 20th century for analysis of commercial catch data. Virtual population analysis (VPA) and catch-at-age analysis were the commonly used methods at the time, making use of age-based data. These models were fully deterministic, without an error assumption, since the models were meant to predict exactly the observed catch. These models relied on a guess to initiate the parameter estimation procedure, and the use of catch data alone often would not contain enough information to properly estimate the fishing mortality in the most recent year (Megrey, 1988). From 1980's onwards, research efforts concentrated on developing methods that included some source of fishery-independent data like survey data collected by research vessels (Megrey, 1988). With the widespread availability of high-speed computing resources, stock assessment models became more complex but far more flexible in the way the models were formulated with less restrictive assumptions. Contemporary assessments can integrate multiple sources of data (commercial CPUE, age and/or length compositions, survey indices, tagging data) and account for all kinds of uncertainty both in the data and in the model. Nowadays, the decision on which model to use depends on what type of data is available.

The Strategic Initiative for Stock Assessment Methods has classified stock assessment models into 8 categories, according to the amounts or types of data, and the degree of age-structured population dynamics in the model (ICES, 2012). From the simplest (that has less data requirements) to the most complex ("data-rich" assessment), these are: catch-only models; time-series models; biomass dynamics models (or surplus production models); delay-difference models; age-structured production models; VPA based approaches; statistical catch-at-age models; and integrated analysis models. International research efforts are now at the level of providing good practices guidelines on using these assessment models (Punt, 2023; good practices workshops from CAPAM 2022).

For the purpose of this thesis, two main types of models will be considered in detail here: surplus production models and statistical catch-at-age (SCAA) models. Both models are considered in a modern framework of state-space model formulation. This consists of two sub-models: a process model for unobserved quantities (e.g. true stock abundances, since the actual size of a fish stock is unobservable) and an observation model for observed quantities, that relate the data (e.g. survey or catch data) with the process model. This hierarchical structure allows for quantities that are unobserved to be random variables with a specified probability distribution (ICES, 2020b).

Surplus production models, also called biomass dynamics models, aggregate the population into a single biomass unit (in numbers or weight) and do not explicitly include biological characteristics such as age or length-structure, maturity or natural mortality. The "surplus production" represents the increase in population biomass in the absence of fishing, or the amount of catch that can be taken while maintaining the biomass at a constant size (Hilborn and Waters, 1992). One example is the Schaefer model, that describes the biomass trajectory over time, depending on a few parameters such as the intrinsic growth rate of the population, the carrying capacity (or initial biomass if known) and a catchability coefficient related to fishing mortality. The main feature of the Schaefer model is the symmetric relationship between surplus production and population biomass. Surplus production is zero at a biomass of zero, and at a biomass corresponding maximum carrying capacity of the population's environment, which limits growth and reproduction (Hilborn and Waters, 1992). A stock-recruitment relationship, which describes the relationship between the spawning adults and their expected number of recruits, is implicit in the logistic growth of the Schaefer population model. The downside of surplus production models is that they require good contrast in the time-series of data to be able to estimate the parameters, meaning distinct phases of increase or decrease in stock abundance or fishing effort (Hilborn and Waters, 1992). Nonetheless, surplus production models provide a simple method that attempts to describe the population dynamics based on fitting the model to one (or more) time-series of an index of relative abundance (e.g. survey data) and an associated time-series of catch data (Haddon, 2021), with few data requirements when compared with age-structured models. Modern versions of surplus production models include a surplus production model in continuous time (SPiCT; Pedersen and Berg, 2017), and a Schaefer model that accounts for increases in fishing power, set in a Bayesian probabilistic framework (Cook et al., 2021).

SCAA models generally depend on age compositions directly estimated from fishery or survey data, commonly using age-length keys. On the other hand, Integrated Analysis (IA) can use either length or age data, and attempts to integrate data in an unprocessed format (i.e., without using age-length keys). Nonetheless, both model types are based on the fact that they split the population into age groups or cohorts (fish born at the same time) and follow fish cohorts through time subject to exponential decay from total mortality. Total mortality defines the rate of decrease of the population size. The total mortality is decomposed into a mortality component derived from the catches (fishing mortality) and a mortality from all other sources (natural mortality). SCAA models are relatively straightforward to use as opposed to the more complex IA models such as Stock Synthesis (Methot and Wetzel, 2013) and CASAL (C++ algorithmic stock assessment laboratory; Bull et al., 2012). One weakness of the SCAA models is that usually no stock-recruitment dynamics are embedded within the model, so it might need an ad hoc analysis to calculate reference points (ICES, 2012). The most widely used SCAA model in the ICES (International Council for the Exploration of the Sea) context is the State-space Assessment Model (SAM) from Nielsen and Berg (2014). Other models are ASAP (Age Structured Assessment Program; Legault and Restrepo, 1998), TSA (Time Series Analysis; Gudmundsson, 1994; Fryer, 2002), a4a model (Assessment For All from the Joint Research Centre of the European Commission; Jardim et al., 2017) and state-space models set up in a Bayesian parameter estimation framework (Cook, 2019b; Miller and Meyer, 2000; Cook et al., 2015).

It is also possible to estimate stock status and fishing mortality on a relative scale without using information from commercial fisheries, when for example catch data are unreliable. "Survey-only" models can offer a complementary tool that uses exclusively research vessel survey data to estimate management quantities of interest like spawning stock biomass and fishing mortality rate (Cook, 1997; Cook, 2013; SURBAR, from Needle, 2015).

There is a growing demand for stepping away from single-species assessment models and move to the use of ecosystem-based models. While ecosystem models provide a more realistic representation of the system and processes being studied, with fishing impacts better represented across different trophic levels, these are very complex models with high data requirements for parameterisation and fitting of the model. The focus of this thesis will remain on single species assessments as generally used for fisheries management advice in European and UK assessments.

1.3.2 ICES assessments for West of Scotland and Irish sea The International Council for the Exploration of the Sea (ICES) is the scientific body responsible for carrying out stock assessments in the Northeast Atlantic Ocean and adjacent seas. The assessment of the state of commercially relevant fish stocks in ICES is usually performed at a much larger spatial scale than the Firth of Clyde. It encompasses the whole Scottish West Coast, known also as Division 6a. The assessments outputs are summarized as indicators of stock status relative to reference points. Important management quantities are spawning stock biomass (measure of adult fish capable of reproducing), recruitment (measure of the number of young fish entering the population each year), catches and fishing mortality rate. ICES refers to two types of reference points: precautionary approach (PA) reference points and maximum sustainable yield (MSY) reference points (ICES, 2017a). The precautionary approach tries to assure that the stock is within safe biological limits, meaning within boundaries that will not impair stock reproduction and recruitment (also known as recruitment overfishing). It uses reference points related with spawning stock size. For example, B_{lim} corresponds to a deterministic biomass limit below which a stock is considered to have reduced reproductive capacity. The maximum sustainable yield (MSY) is based on harvesting the stock at optimal levels to maximise yields in the long-term, and its reference points are directed towards managing fishing mortality. The concept of MSY relies on the “surplus production” generated by a stock that is being harvested below its carrying capacity (Hilborn and Walters, 1992). The most used reference point is F_{MSY}, which corresponds to the fishing mortality rate expected to give MSY in the long term. When the MSY advice rule is applied, F_{MSY} is constrained so that the long-term probability of spawning biomass falling below B_{lim} is less than 5%.

A stock-recruitment relationship forms the basis for the calculation of these reference points. Stock-recruitment analysis normally consists of looking at the empirical relationship between spawning stock biomass and the subsequent recruitment of the year class produced by that spawning stock (Hilborn and Waters, 1992). Most regularly used models to describe stock-recruitment dynamics are the Beverton-Holt model and the Ricker model (Ricker, 1954; Beverton and Holt, 1957). The main difference between these two models is that while the Beverton-Holt assumes that recruitment increases with spawning biomass until it reaches an asymptote, the Ricker curve does not attain an asymptotic limit but instead assumes a decline in recruitment levels at high values of spawning stock biomass. Various mechanisms have been proposed to explain this decline: cannibalism of juveniles by adults, density-dependent transmission of disease, and density-dependent growth combined with size-dependent predation (Haddon, 2021). The selection of an appropriate model is crucial. Often recruitment is highly variable, sporadic, or with limited stock-recruitment data, which makes it difficult to identify the shape of the stock-recruitment relationships. Fluctuations in environmental conditions contribute to recruitment variability from year to year. In most ICES assessment models, recruitment is considered as a random walk process to account for this variability. Modelling the recruitment process also forms the basis for forward projections of the stock and providing advice of alternative management strategies. Within ICES, a stochastic equilibrium software tool ("EqSim") is used to estimate the stock-recruitment dynamics, project the stock forward for the next 200 years or until reaches an equilibrium state, and to calculate biological reference points (ICES, 2019).

The ICES assessments for Scottish waters are conducted for a number of demersal, pelagic and shellfish species. The full list of assessed stocks for the Celtic Seas ecoregion can be found in ICES (2022). Here the focus will be on three demersal fish species that are relevant at the level of the Firth of Clyde – cod, haddock and whiting – in the West of Scotland (division 6.a) and Irish Sea (division 7.a).

Whiting is assessed separately for the West of Scotland and the Irish Sea. For 2023, ICES recommends zero catches in the Irish Sea, and a small (compared to historical values) catch for the West of Scotland, the first non-zero advice in five consecutive years (ICES, 2022a; ICES, 2022b). It seems that spawning stock biomass has recovered slightly for whiting in 6a since 2014, with recruitment varying around low levels. Fishing mortality declined almost continuously since 2000 and has been below F_{MSY} since 2005. In contrast, the spawning biomass of whiting in 7a has been below reference points since mid-1990s, with recruitment being low since the same period. Fishing mortality remains above F_{MSY} since the beginning of the assessment in 1980. The size of the whiting stock in 7a is estimated to be very low and is primarily taken as bycatch within the Nephrops fishery.

The size of the cod stock in both 6a and 7a is extremely reduced, and zero catch advice is given for both areas for 2023, although the advice has changed in 2024 (ICES, 2022a; ICES, 2022b). The cod stock in 6a has had zero catch advice since 2004. A stock identification workshop concluded there are multiple subpopulations in 6a, a subpopulation in offshore areas that is connected to the North Sea and a separate subpopulations in inshore areas, one of them in the Clyde with a connection to the Irish Sea (ICES, 2022d). In 2022, the assessment was conducted at the level of 6a division. A benchmark workshop in 2023 combined the North Sea and West of Scotland cod stocks into one single substock unit, called Northwestern stock (ICES, 2023a; ICES, 2023b). In the 2022 assessment, the spawning stock biomass was well below reference points, and fishing mortality above F_{MSY} even though there has been a clear decrease in fishing mortality since 2009. The advice for cod in 7a is similar to 6a, but with a difference that fishing mortality has been below F_{MSY} since 2010. However, spawning biomass has declined sharply since the beginning of the assessment around 1970, and recruitment has been at historically low levels since mid-1990s. A recreational fishery for cod in the Irish Sea is considered to take up similar levels of catches as the commercial fishery, and an alternative reference point for fishing mortality is provided, based on environmental data (F_{ECO}). The MSY approach is questioned in the cod 7a assessment, since it seems that recruitment rather than fishing pressure is driving stock trends.

Haddock is assessed at a larger scale than the west of Scotland. The assessment for haddock was changed in 2014, grouping the stocks from the North Sea and the West of Scotland as a single population (known as Northern Shelf haddock; ICES, 2014). The stock is considered to be within safe boundaries, with fishing pressure below F_{MSY}, and the spawning stock size above limit reference points (ICES, 2022c). There had been recently two strong year classes (2019 and 2020) that entered the population from sporadic successful recruitment events, which produced a sharp increase in SSB at historically high values and will continue to impact the catch advice in the upcoming years. In the Irish Sea, the haddock stock is analysed separately (ICES, 2022b). Since 2014, there has been a slight increase in the activity of the whitefish fishery in the area due to a rapid increase in abundance of the haddock stock. Fishing pressure is below F_{MSY}, and spawning stock size is above limit reference points. The haddock stock is characterized by highly variable recruitment, with sporadic peaks of strong year classes. Recruitment in 2013 is amongst the highest observed since 1990 and was followed by strong recruitment in 2014 and 2015. Spawning stock is now declining after reaching its highest historical level in 2018.

There are a few assessments conducted at the level of the Firth of Clyde. The Clyde herring stock was assessed separately from the 6a area before the stock collapsed in 1970s (McIntyre et al., 2012). The Nephrops stocks are managed by functional units, which defined discrete patches of mud which they inhabit. The Clyde Nephrops population is assessed separately from the populations in the wider west of Scotland but together with the Sound of Jura (ICES, 2022a). Nephrops in the Clyde occurs at higher densities when compared with other functional units, suggesting a relatively high productivity. Historical harvest rates in the Clyde have been generally high, at or above reference points.

There are no assessments of demersal fish species done at the Clyde level. However, the ICES workshop in stock identification of West of Scotland Sea cod (ICES, 2022d) clearly identifies the need of separating the Clyde cod stock from the rest of the 6a area. The benchmark workshop on Northern Haddock stocks (ICES, 2014) also presents evidence of a somewhat isolated haddock population within the Clyde. Before treating the Clyde demersal fish populations as isolated units, a review of the existent literature on the connectivity between the Clyde and adjacent areas is conducted in the next section.

1.3.3 Stock structure and connection between Clyde and adjacent areas It is now widely acknowledged that many stock assessment areas do not reflect boundaries of fish populations (Reiss et al., 2009). A fish stock considered in fish stock assessments is not always equivalent to a fish population in biological terms. A unit stock is a management quantity that considers a closed population (no emigration or immigration) of individuals of one species that have the same life history, growth rates and spawning grounds. Population connectivity can lead to individuals moving between discrete populations and genetic connectivity can allow for gene flow. Demographic connectivity in marine fish populations can occur at different life stages, such as egg dispersal, ontogenetic shifts from juvenile to mature stages, or the seasonal spawning migrations of adults (Burns et al., 2020). These widescale movements pose a challenge on the definition of management areas and boundaries.

For the gadoid species of interest in this work, there is an extensive body of work on spawning grounds and metapopulation structure in North Atlantic waters. Several studies demonstrated that cod stocks in the West of Scotland are structured in relatively closed subpopulations, with the majority of spawning adults originating from resident areas (Wright et al., 2006a; Wright et al., 2006b; Gibb et al., 2007; Galley et al., 2006). In North Atlantic waters, cod aggregate in large numbers to spawn, between February and June. Pelagic eggs give rise to larvae after hatching and these metamorphose into juvenile fish. The juveniles settle onto banks or coastal areas (Ryan and Bailey, 2012). Young fish move out of shallow coastal waters during the first winter, and gradually move to deeper waters as they age to join adult populations (Bailey et al., 2011). Structurally complex habitats in shallow waters are important to juvenile cod, likely reducing predation risk. Early in their life history, cod juveniles feed on zooplankton and benthic prey, and then switch to a piscivorous diet when reaching adult stage. Cannibalism occurs when prey fish reach 1/3 the length of the predator, and this behaviour has important implications for the dynamics of cod populations and for survival in habitats chosen by the juveniles (Grant and Brown, 1998). The report from ICES (2022c) collects all literature on stock structure and identification from tag-recapture studies, otolith microchemistry, otolith shape analysis and genetic data. It concludes that most inshore spawning groups are characterized by a high level of residency, relied on local recruitment, and showed little to no mixing with neighbouring areas, even at a small spatial scale. However, offshore groups to the west of the Outer Hebrides and other offshore areas in 6a showed a larger extent of movements and mixing with groups in Shetland and the Northeast (division 4a, north of North Sea). The most likely hypothesis in terms of population structure in the West of Scotland is having multiple overlapping subpopulations related to the Dogger stocks (south-central North Sea), connecting the northwest of Scotland to the North Sea (between 6a and 4a divisions), and a separate subpopulation in the Clyde. There is evidence that cod in the Clyde can be part of a wider Celtic unit, encompassing the Irish Sea, Celtic Sea and western Channel (Heath et al., 2014), with limited exchange of individuals between the Clyde and the open shelf in west of Scotland.

In contrast, whiting and haddock show evidence of wider dispersion and connectivity across the West of Scotland. Haddock has a widespread range across the west coast of Scotland and North Sea, but from mid-1990s onwards there was a marked expansion on its distribution and abundance across the Irish and the Celtic Seas (Dickey-Collas et al., 2003). The ICES (2014) report concludes there is biological justification for combining the haddock stocks from the North Sea, Skagerrak and West of Scotland into a single unit (Northern Shelf) since there is enough connectivity at early life stages (larvae and juveniles) between the regions. Haddock spawns from January to May, forming large aggregations and producing large quantities of pelagic eggs that float at the surface, until hatching followed by the larval stage. This phase can take up to 6 months, during which eggs and larvae are likely to be passively transported (Bailey et al., 2011). Much of the egg production from the Scottish west coast may be advected to the North Sea (Heath and Gallego, 1997). There is also evidence of mixed origins of adult haddock, indicating a substantial juvenile dispersal before settlement into a demersal life stage (Wright et al., 2010). However, there seems to be a relatively low exchange of mature adult stock between West of Scotland and North Sea, that could lead to locally different fishing mortality rates (ICES, 2014). Most adult haddock in the Firth of Clyde had the otolith micro-chemistry signature of the local juvenile nursery area, giving support to the hypothesis that haddock in the Clyde are separated from the wider west coast of Scotland (ICES, 2014). In the Irish Sea, there is a persistent reproductive isolated haddock population, that could be connected to some extent with the Firth of Clyde but further studies are needed to confirm this (ICES, 2014).

Whiting spawns between February and June, and after 2-3 months, the eggs hatch into pelagic larvae that undergo through metamorphosis and a short pelagic juvenile phase. Juvenile whiting settle in nearshore waters between June and December, and then move offshore as larger age-1 fish for the following winter period (Bailet et al., 2011). Contrary to other gadoid species, immature and mature whiting are found together during spawning season suggesting this species does not exhibit spatially distinct spawning areas (Burns et al., 2019). Tagging studies show that most adult whiting do not travel long distances, with little exchange of individuals between the west of Scotland and the North sea. However, otolith elemental analysis indicated that there is extensive active dispersal of juveniles following settlement (Tobin et al., 2010). Recent work from Burns et al. (2019, 2020) indicated that movements of juvenile to adult stage whiting connect most inshore waters of the west of Scotland to the eastern Irish Sea. Age-1 whiting in the inshore waters of west of Scotland are likely to be derived from age-0 fish from the Clyde, while age-1 whiting in the Clyde are most likely recruited from local age-0 whiting. There seems to be some connectivity between the eastern Irish Sea and the Firth of Clyde, with a northerly movement of young juveniles from the Irish Sea to the Clyde region. From settlement to spawning, there are substantial ontogenetic distribution shifts in whiting that result in a movement from coastal to offshore waters as fish grow older (Burns et al., 2019).

Overall, the only species for which there is clear evidence of an isolated population at the Clyde level is cod. For haddock, even though there is some separation between Clyde and the wider 6a division, the Clyde stock might be connected to the Irish Sea. There is a clear exchange of young whiting between Clyde and the Irish Sea, and then an offshore migration to deeper waters of adult whiting into the wider Scottish west coast. For the purposes of this work, the Clyde whitefish stocks are treated as closed populations, and the implications of this assumption are discussed.

Thesis aims and objectives

This thesis investigates the hypothesis that bycatch in the Nephrops fishery is preventing the recovery of whitefish stocks in the Clyde, namely cod, haddock and whiting. To test this hypothesis, a number of sequential tasks and objectives were undertaken. First, all relevant data for demersal fish species in the Firth of Clyde was extracted and compiled, both fishery-independent data (research vessels surveys) and fishery-dependent information (observer and logbook data). Total quantities of fish discards in the Clyde Nephrops fishery were estimated using different methodologies and drivers of discarded quantities investigated. Then, stock assessment models were developed to estimate mortality rates and population abundance, and set in a Bayesian parameter estimation framework. Model simulations with different scenarios tested how estimated mortality rates are affecting the recovery of the populations. These outputs allow a greater understanding of the effect of discarding on the health, status and productivity of demersal fish species in the Clyde region, and will help to advance knowledge on the management of the fisheries in this area.

Thesis overview

Chapter 2 presents the data extraction and processing steps conducted for the three sources of data: observer data, logbook data and survey data. It also gives a brief account on the sampling schemes and design involved in collecting these datasets. The processed discards and survey data from Chapter 2 are used in all subsequent Chapters in the following manner (see Figure 1.1 for a flow diagram with the data processing steps explained Chapter 2 and the overall thesis structure). In Chapter 3, estimates of discards from the observer trips are raised at the fleet level using different statistical approaches, based on the discard weight per trip and information at the fishing fleet level (survey data is also included). Chapter 4 presents an age-structured model (ASM) that includes the discards/catch data in numbers of fish at age for the fleet (from Chapter 3), and the survey data as numbers of fish per hour of sampling effort. Model testing and validation were conducted using simulated data and data for the West of Scotland. In Chapter 5, the ASM is applied to the main demersal fish species in the Clyde (haddock, whiting and cod). Sensitivity analysis guided the choice of the most appropriate model version for each stock. Chapter 6 explores an alternative modelling approach using a surplus production model. This model uses only the discard weight and the survey data as total weight per hour of trawling, not including age compositions. For Chapter 7, the outputs from the ASM are used to project the fish populations into the future under different mortality scenarios. Reference points commonly used in fisheries management are estimated as a basis for determining the probability of recovery of the Clyde stocks. Chapter 8 provides an overall discussion of the results from all previous Chapters, answers the goals of the project and summarizes main conclusions, while acknowledging the limitations of the work and pointing further considerations of interest to the topic.

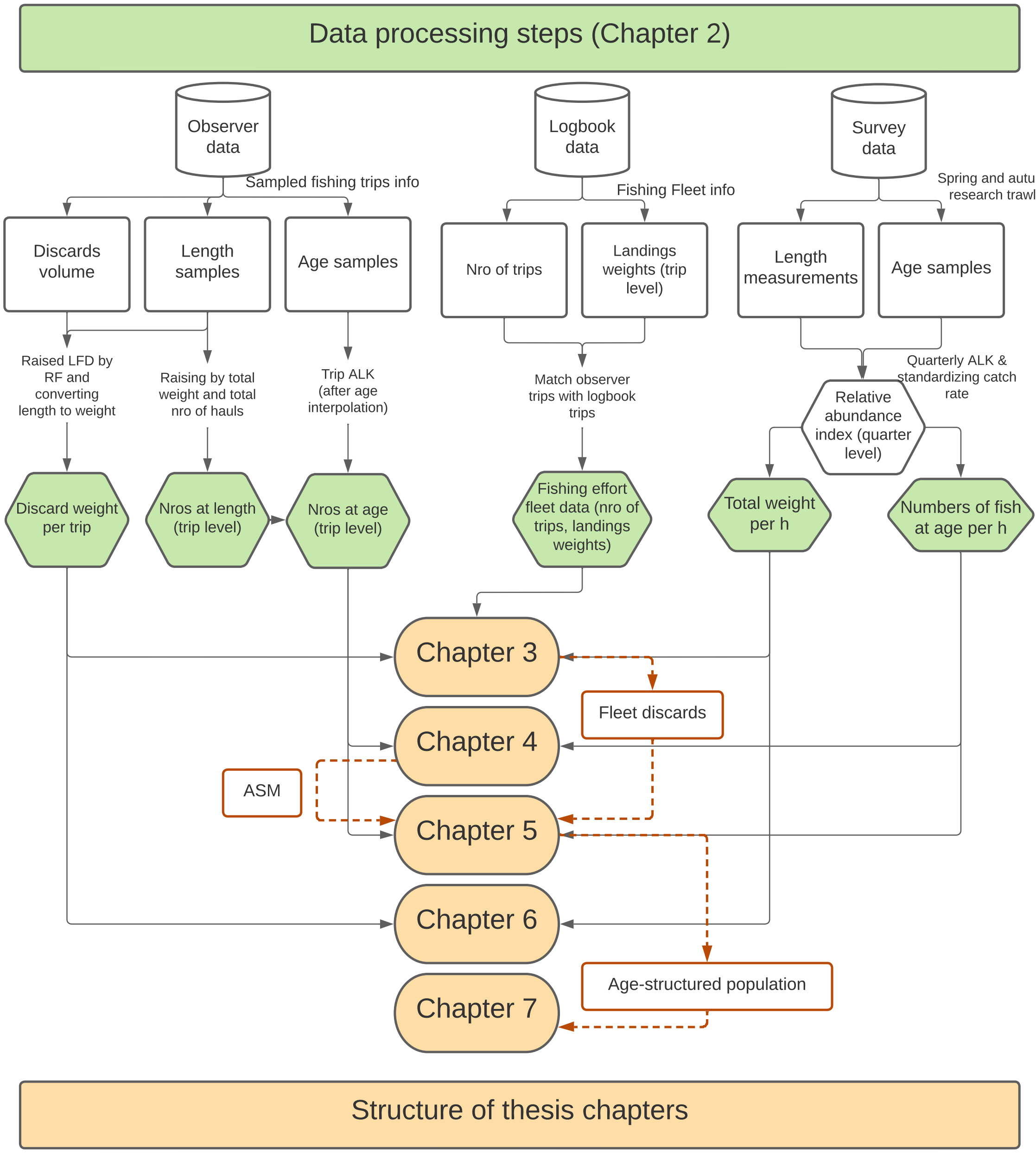

Figure 1.1: Structure of the thesis highlighting the data processing steps conducted in Chapter 2 and how these link with each subsequent chapters. LFD - length-frequency distribution; RF - raising factor; ALK - age-length key; ASM - age-structured model.

CHAPTER 2 - Assembling available data for the Clyde

2.1 Introduction

Data for demersal fish species in the Firth of Clyde comprise fisheries-dependent data and fisheries-independent data. The fisheries-dependent data consisted of information collected by observers on-board commercial fishing vessels (“observer data”) and information officially reported by the fishermen when arriving at port (“logbook data”). The fisheries-independent data are collected by Marine Scotland Science following a standardized protocol with research vessels (“survey data”). The observer data and logbook databases were accessed through a direct collaboration with Marine Scotland Science, while the survey data are publicly available. The ICES statistical rectangles were used as a basis for extracting the data sets (40E4, 40E5, 39E4, 39E5; Figure 2.1), as these form the smallest area unit for officially reported landings in the Clyde. These include a small area outside of the Firth of Clyde (Sound of Jura and the North Channel). The data were used to calculate indices of relative abundance (from the scientific surveys, e.g. weight of fish caught per hour of sampling) and catch/discards in numbers of fish per age class, fundamental inputs for analytical stock assessments that are developed in the subsequent Chapters.

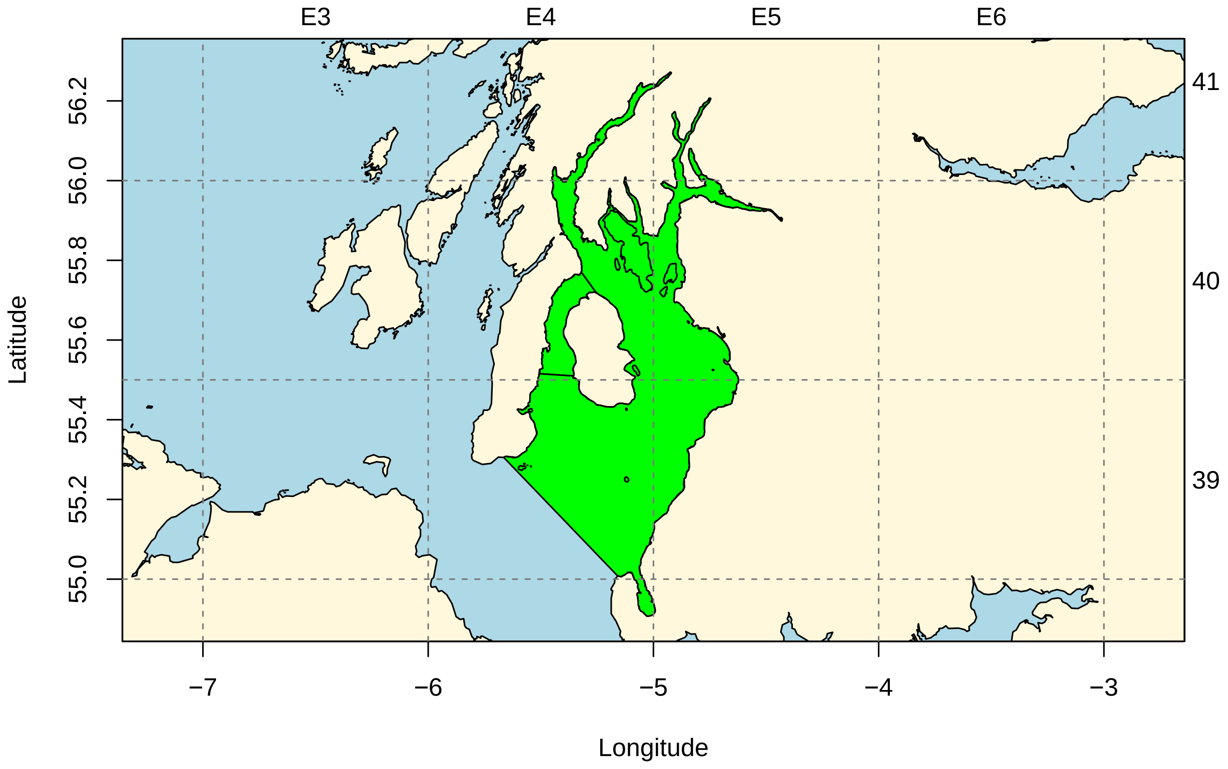

Figure 2.1: Firth of Clyde (green area) and ICES statistical rectangles defined by row numbers (right) and column numbers (top).

2.2 Methods

2.2.1 Observer data

The discard sampling programme onboard Nephrops trawlers started in 1982 in the Firth of Clyde, focused on sampling the main commercial species such as herring (*Clupea harengus*), cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangius*) and saithe (*Pollachius virens*). Observers are placed onboard commercial vessels chosen at random to collect information at sea on discards and landings over consecutive days. The participation of the vessels in the sampling programme is voluntary. The whole of the West of Scotland (ICES Division 6a) is stratified into 9 subunits with 20 to 30 trips sampled per year. However, the number of trips taking place in the Firth of Clyde is low given its small size in relation to 6a (Figure A.1 in Appendix A - Supplementary Info). There is no fish bycatch data from creel vessels being collected at present; also, fish bycatch in creels is believed to be negligible.

For each sampled trip, which is defined here as the time between the vessel leaving the harbour and arriving back to land the catch, the observer collects detailed information about the fishing activity (gear type, mesh size, location, date and time, landed weights). For each haul, a sample of 1 basket of fish with a known weight is taken from the total catch. Typically such a sample would weight approximately 35 kg. Each fish in the sample is identified to species level, and measured for length. Otoliths for age-determination are extracted from a subsample of fish. Since the subsampling of biological information (length, weight, age and maturity stages) occurs in three nested steps (trip -> haul -> basket), these need to be raised to the trip level first. The age data are collected as a length-stratified subsample of fish measured for length.

For each haul and species, the length-frequency distribution (LFD) is defined as the number of sampled fish per length class. The LFD is converted to a sampled weight using species-specific length-weight relationships defined by

$$W = aL^b \tag{2.1}$$

where W is weight in grams, L is the length class in cm, and a and b are parameters with values taken from Coull et al. (1989). The parameters used are available in Table A.1 in Appendix A (Supplementary Information). For the observer data, values of a were chosen from June as the midpoint of the year.

The sampled discard weight for each species is calculated using the LFD as

$$W_{haul} = G\ a \sum_{i=1}^{n} N_i\ L_i^b \tag{2.2}$$

where W_{haul} is the weight of fish in each haul, G is a factor to convert gutted weight to total weight of a fish, n is the number of length classes, N_i are the number of fish in length class i in the sample, L_i the length of fish in length class i, and a and b are the parameters of the length-weight relationship.

The total weight of fish discarded for each haul is approximated as the total number of baskets of fish discards (unsampled), so a raising factor is applied to the sampled weight:

$$Total\ W_{haul} = Sampled\ W_{haul} \frac{total\ number\ of\ baskets}{sampled\ number\ of\ baskets} \tag{2.3}$$

By summing the total weight across hauls, we obtain the total weight of fish discards for each trip ($d_{t,y}$, with t representing trip and y a subscript for year; this notation will be kept throughout Chapters 2 and 3).

To calculate the number of fish in each age class, age-length keys (ALKs), derived from otolith samples, were used for each trip and applied to the LFDs. Since there were few trips in the Clyde, age and length data for each trip had to be carefully analysed so as to include as much data as possible. Trips for which there were no otoliths collected were excluded from further analysis. For trips with lengths but missing a corresponding age class in the ALK, age interpolation methods were used to fill the gaps. A cubic smoothing spline was fitted to interpolate missing ages for each trip ALK (Hastie and Tibshirani, 1990). A smoothing spline allows for a high degree of flexibility when fitting the model to the data points, while accounting for a penalizing term that handles the smoothness of the curve (Perperoglou et al., 2019). In a few cases where the spline gave unrealistic results (negative ages or extreme values), a mean age method was applied, that uses the mean age of the nearest length group to fill out missing ages (considering the bin size of the length group 3 cm).

The ALKs are representative of a length-stratified subsample of the length compositions (equation A.1 in Appendix A). To obtain the proportion of fish at length given a certain age for each trip, the ALKs were multiplied by the raised LFDs. The numbers of fish at age and length for each trip were summed across length classes to obtain number of fish at age for each trip ($n_{t,a}$).

Age information from the observer databases was available between 2002 and 2019, making a total of 118 sampled trips. Discards data was extracted for the Firth of Clyde using trip coordinates between 2008 - 2019. For data prior to 2008, there were no latitude/longitude digitized data so information at the statistical rectangle level had to be used as a spatial criterion. This included a small area outside the Clyde (west of the Kintyre peninsula; statistical squares 39E4 and 40E4, Figure 2.1). Spatial plots showed very few observer trips took place in this area outside the Clyde (Figure 2.2), so including these observer trips had only a small contribution to the total discards calculated.

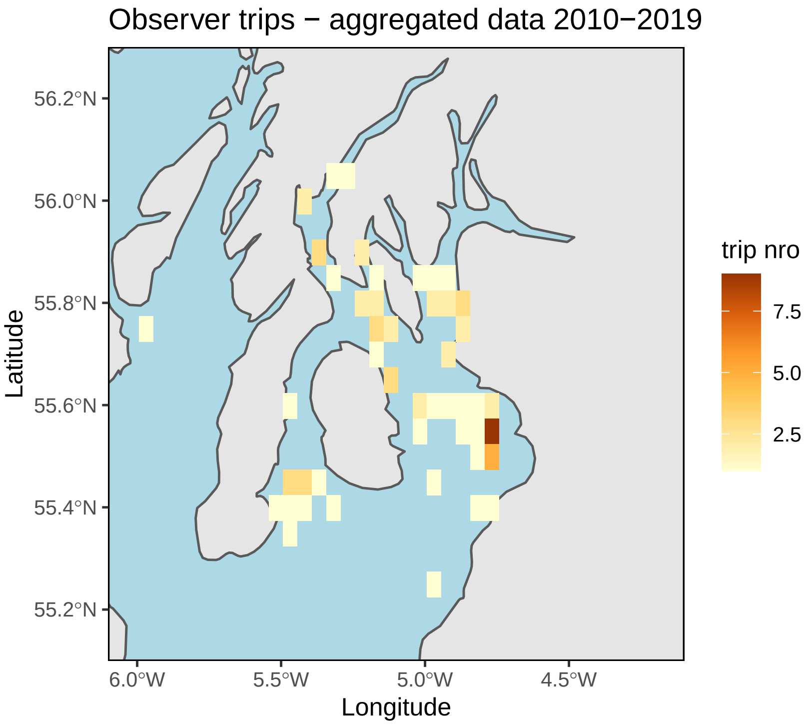

Figure 2.2: Observer trips location in the Clyde. Aggregated data from 2010 to 2019. Trip nro represents total number of trips.

By-catch weight of various demersal fish species was available after 2007: flatfish species such as witch (*Glyptocephalus cynoglossus*), dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*); other gadoids like hake (*Merluccius merluccius*) and Norway pout (*Trisopterus esmarkii*); and lesser-spotted dogfish (*Scyliorhinus canicula*). However, age data had been collected continuously only for cod, haddock and whiting, so the analysis focused on these three main commercial species. For the period between 2002 and 2019, there were 3 trips with zero discards only for cod. These trips were excluded from further analysis.

Data for observer trips from 1990-2001 was also available. However, this historical data did not contain age information, only discarded weight, so the numbers of fish discarded for each age class could not be estimated. Additionally, there was no information on Nephrops landed weights corresponding to the observer trips, and this is an important variable that will be used to raise the observer data to the Nephrops fleet level, since Nephrops landings will be used as a proxy for fishing effort. This is explained in more detail on the following section.

2.2.2 Nephrops landings as proxy for fishing effort

Discards for the Scottish fleet are currently estimated using a stratified ratio estimator, with demersal fish landings as an auxiliary variable (Stratoudakis et al., 2001). However, there are almost no fish landings in the Clyde for the period of data available, so an alternative variable to be investigated is quantities of Nephrops landed. Although there is information on the hours fished for the sampled trips, this information is not available for the trips at the fleet level. The Nephrops landed weights per observer trip (l_t) might be used as a proxy for hours fished, assuming that the longer the vessel fishes for, more Nephrops it will catch. If B_{nep} represents the biomass of Nephrops in the sea and h is a measure of time fishing then:

$$l_t \propto h \; B_{nep} \tag{2.4}$$

and by arrangement

$$h \propto l_t / B_{nep} \tag{2.5}$$

Clearly if B_{nep} remains relatively constant over time, then h is proportional to l_t and is a proxy for effort. Considering that Nephrops abundance from underwater TV surveys represents a measure of Nephrops biomass, and after checking that the biomass of Nephrops has little long time trend since 2002 (Figure A.3 in Appendix A), we assume that the Nephrops landings are an approximate measure of hours fished:

$$h = c \; l_t \tag{2.6}$$

and c is a constant of proportionality.

2.2.3 Logbook data

The logbook data consists of information on each trip collected by fishers during their fishing activities. It contains the type of fishing gear (gear code and mesh size), area (main fishing area and ICES statistical rectangle), vessel identification information, start and end of fishing trip, landing date and port, flag of the country, species and quantities landed and value of the landings. The annual number of fishing trips and Nephrops landed weights for the Clyde Nephrops fleet were extracted from this source. All landings occurring within the Firth of Clyde were used, both from UK vessels and Northern Irish vessels, for bottom otter trawls (OTB) and multi-rig otter trawls (OTT) targeting Norway lobster. The Clyde landings were extracted using the ICES statistical rectangle (no coordinates were available), which includes the area to the west of Kintyre (which represents a small portion of the total landings).

One sampled trip is considered as the time between the vessel leaving port and arriving back to land the catch. To calculate the total number of trips for the fleet (and associated landings), the observer trips had to be matched with the logbook trips using vessel identification numbers and landing dates. This processing step allowed the correction of apparent errors in Nephrops landed weights for the observer trips, since the duration of one observer trip is not the same as one logbook trip; meaning that one observer trip could cover several landings made by one vessel (e.g. 2 logbook entries might match 1 observer trip over consecutive days). This raises a potential issue that one observer trip can be different from one logbook trip.

Logbook data was available from 1985 until 2019, with different levels of detail at separate time frames. Two subsets of data were extracted. The first had data between 2002 and 2019, which contained detailed information at the trip level, such as vessel identification information and trip date, so that could be matched with the observer trips. Previous to 2002, this level of detail was not available. The second data set comprised data between 1985 and 2002, and contained Nephrops landed weights and fish landed weights by statistical rectangle since there were no coordinates available. Fish landings (cod, haddock and whiting) were low in 2002 and decreased to almost non-existent from 2008 onward (Figure A.2 in Appendix A).

2.2.4 Survey data

Data for the Firth of Clyde was gathered from the International Bottom Trawl Survey for the Scottish West Coast (SWC-IBTS; 1985 – 2011) and the West Coast Ground Fish Survey (SCOWCGFS; 2011 – 2019), and complemented with a few samples from the Irish Ground Fish Survey (IGFS; 1996 – 2018) and bottom trawl samples from the Fisheries Research Services (FRS) historical surveys to fill up year-quarter gaps. Trawl survey data is openly accessible and available to download from the ICES website (https://datras.ices.dk/Data_products/Download/Download_Data_public.aspx). This data was collected following a standardized protocol with a Grande Overture Vertical (GOV) otter trawl during quarter 1 (January – March) and quarter 4 (October – December) each year. In 2011, a new survey design for the West Coast of Scotland was in place, changing from a fixed station design to a randomised stratified survey. A change in ground gear design was required to be capable of working on hard substrates that might be present in random allocated stations, and as a consequence this can affect the survey efficiency of catching fish (ICES, 2017a). In the same year, trawl duration changed as well from 60 min to 30 min tows. Detailed information for each haul was registered (exact location - Figure 2.3, trawl duration, date, wingspread) and all fish or random samples of fish caught are identified, counted and measured to the nearest centimetre below.

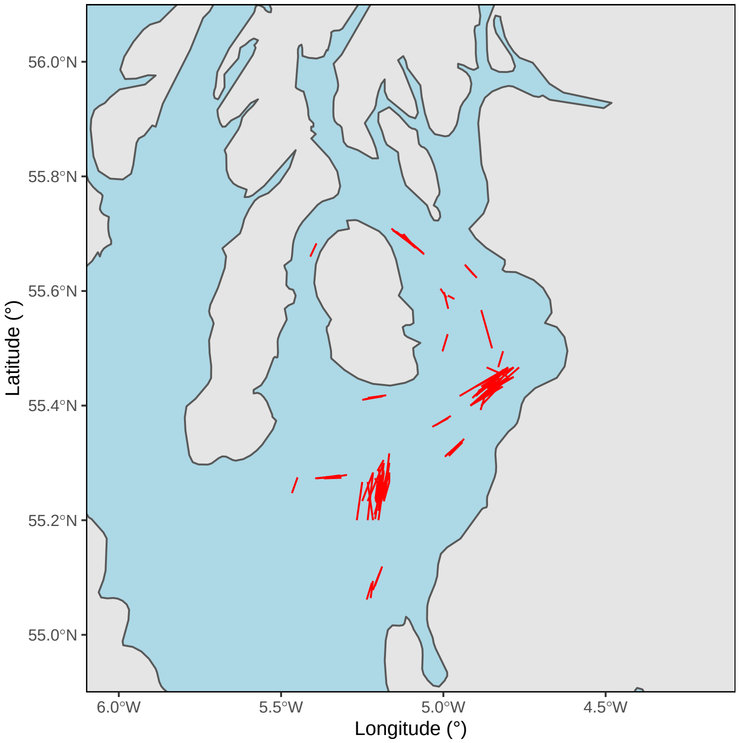

Figure 2.3: Survey sampling locations in the Clyde. Data from 1985 to 2019, aggregated for Q1 and Q4 surveys.

To obtain a relative index of fish abundance (in weight), the numbers of fish in each length class (i.e. length-frequency distributions, LFD) were calculated first, aggregating data across hauls separately for Q1 and Q4 surveys, and standardized by trawl duration (total number of hours trawled per quarter). A biomass index was first calculated, by converting the numbers to weight-at-length using length-weight relationships, and summed across lengths to obtain total weight for each quarter. The length-weight relationships used are shown in Table A.1 (Appendix A) and were extracted from Coull et al. (1989). Value of a corresponding to March was used for the quarter 1 (Q1) survey, and a value of a corresponding to November was used for the quarter 4 (Q4) survey.

The survey data contains detailed biological data such as length, age and maturity of fish. For each haul, a sub-sample is taken for age determination by extraction of otoliths; this creates a length-stratified age data subset. To calculate a relative index of fish abundance as numbers per age class, for each species, the ALK was applied to the length sampled fraction (without age subsampling), separately for each quarter. The numbers of fish were summed across length classes to obtain the numbers of fish at age (see equations A.2-A.3 in Appendix A). The numbers (or weight) of fish at age were standardized per hour of towed gear (using total number of hours of trawling per quarter).

2.2.5 Mean weight at age data

The mean weight of a fish for each age class has to be determined from either the catch or survey data for estimating important stock summary statistics such as spawning biomass. First, the proportions of fish at age and length were converted to weight at age and length distributions, and then averaged across length classes to obtain mean weight at age of an individual fish (equations A.4-A.5 in Appendix A).

The mean weights at age for each species were calculated separately using the discards data, the Q1 survey and the Q4 survey. For the discards data, the proportions of fish at age and length were calculated on a yearly basis (not for each trip). Length-weight relationships specific to each species were applied as per equation 2.1.

2.2.5.1 Smoothed mean weight at age The estimated mean weights at age have unrealistically high variability from year to year. It is not expected that the average weight of one fish, particularly at older ages, would vary so much since growth slows down with age reaching a maximum weight/length. The variability can be explained by the overall low sample size for the Clyde data and there are few fish older than age 2. A simple time series model was applied to the mean weights at age to reduce the variability in the data and fill up gaps in the time series.

The model is summarised as follows:

$$W_{y+1,a} \sim Normal(W_{y,a}, \sigma^W_{y,a}) \tag{2.7}$$

where $\sigma^W_{y,a}$ is the standard deviation. Assuming a constant coefficient of variation cv, across all age groups, the standard deviation is given by:

$$\sigma^W_{y,a} = cv\ W_{y,a} \tag{2.8}$$

The observed mean weights are normally distributed with observation error σ^{obs}_a:

$$W^{'}_{y,a} \sim Normal(W_{y,a}, \sigma^{obs}_a) \tag{2.9}$$

The prior distributions on the parameters when fitting the model using Bayesian methods were:

$$cv \sim Uniform(0, 1) \tag{2.10}$$

$$\sigma^{obs}_a \sim Uniform(0, 100) \tag{2.11}$$

$$W_{1,a} \sim Uniform(0, 50) \tag{2.12}$$

with cv being the coefficient of variation, σ^{obs}_a the observation error and $W_{1,a}$ the mean weight at age for the first year of data.

This model was implemented in the Bayesian package “RStan” (Stan Development Team, 2023) and applied to each species (cod, haddock and whiting). RStan provides an interface between the R software and the STAN probabilistic programming language. RStan samples the posterior distributions with a No-U-Turn sampler (NUTS) given user-specified prior distributions and a likelihood approach given the data. Model convergence was assessed with trace plots that track each MCMC chain, and provide a visual way to inspect sampling behaviour and assess mixing across chains. The potential scale reduction factor, R-hat, measures the ratio of total variability across chains to the within-chain variability. After convergence, the chains should have approximately the same level of variability since the samples should come from the same posterior distribution, so the R-hat should be close to 1. The equations for the posterior distribution and likelihood term are shown in Appendix A (equations A.10 - A.11).

Preliminary runs with 3 chains and 10 000 iterations were assessed with trace plots and use of the R-hat. Final runs consisted of three MCMC chains with 50 000 iterations and a thinning rate of 150. Samples of the posterior distribution were saved for each chain and used to calculate standard statistics including the mean, median and 95% credible intervals for all critical parameters. Histograms of the posterior parameters allowed visualization of the distributions. Where there were skewed distributions, the median values were chosen instead of the mean for further analysis.

The Q1 and Q4 survey weights were modelled separately, and then an average over the two was calculated to give a proxy for mean weight at age over the year. The mean weight at age used in the ICES working groups were extracted from the reports for each species for the west of Scotland (division VIa; ICES, 2013; ICES, 2021a) and compared with these "smooth" weights calculated for the Clyde data. Mean weight at age 0 was considered constant.

2.3 Results and Discussion

Table 2.1 provides a summary per year of the total number of fishing fleet trips (of Nephrops trawlers operating in the Clyde), total number of observer trips, the total weight of Nephrops landed by the fleet and the Nephrops landed weight registered in the observer trips. The number of logbook trips that were matched with the observer trips is also provided. The observer trips represent a low sampling effort of around 0.1% coverage of total fleet trips or Nephrops landed weight. This is similar to the level of sampling observed in other studies (Stratoudakis et al., 2001; Catchpole et al., 2011).

Table 2.1: Total number of fishing fleet trips, number of observer trips, number of logbook trips matched to observer trips (Log trips*), total Nephrops landings weight by the fleet (Nep lands fleet), Nephrops landings weight for observer trips (Nep lands obs).

Year	Fleet trips	Obs trips	Log trips*	Nep lands fleet (tonnes)	Nep lands obs (tonnes)
2002	8592	5	11	1568.9	1.9
2003	9823	4	7	1673.5	1.2
2004	7833	6	7	1487.5	2.0
2005	9020	4	10	1982.9	2.5
2006	10877	5	5	2496.8	0.9
2007	11747	2	5	3229.5	1.1
2008	10263	3	6	2880.5	1.7
2009	10187	4	14	2474.4	2.6
2010	10950	4	7	2995.7	2.1
2011	10611	5	11	3138.8	2.8
2012	10141	3	3	2997.4	1.1
2013	8756	11	16	2273.5	4.2
2014	7397	14	19	2088.3	4.9
2015	6537	10	12	1884.2	3.7
2016	6951	10	10	2486.8	3.2
2017	8864	11	14	2116.0	3.1
2018	7950	2	2	1689.1	0.3
2019	7322	11	12	1708.3	2.1

The mean numbers of fish at age per observer trip were calculated from the observer data bases. Figures 2.4 - 2.6 represent the mean discarded numbers at age for each species. For haddock, the discards are composed of mainly age 0 and age 1 fish, and a lower proportion of age 2 fish. For whiting, discards are constituted by age 0 and age 1 fish. Mean discarded numbers seem to have decreased from 2002 to 2019 for both haddock and whiting. There were very few cod caught by the Nephrops trawlers compared with the scale of values for haddock and whiting. Discarded cod was mostly age 1 fish, and in lower quantities age 0 and age 2 fish. There is no obvious trend in the cod discards across years, probably because of the low numbers of cod caught.

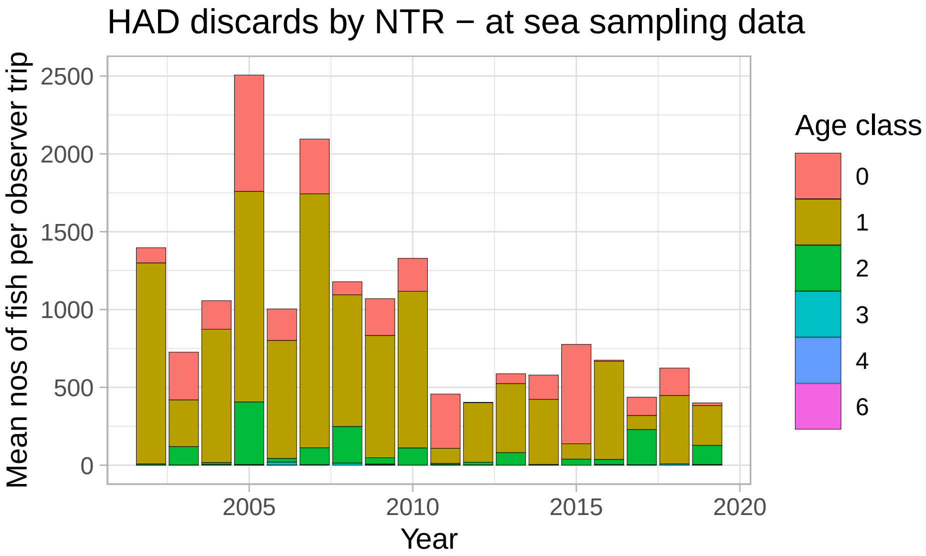

Figure 2.4: Mean numbers of fish at age discarded per observer trip for haddock (HAD). NTR - Nephrops trawlers.

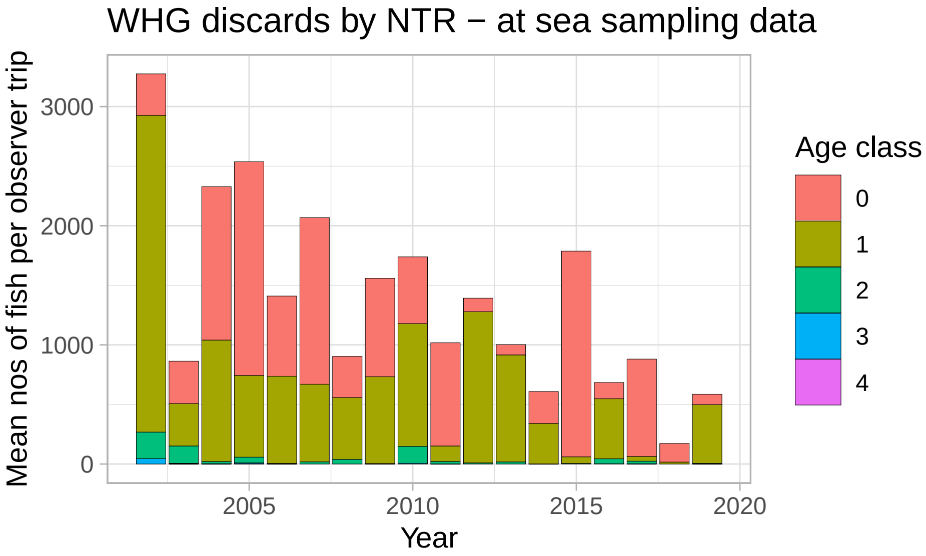

Figure 2.5: Mean numbers of fish at age discarded per observer trip for whiting (WHG). NTR - Nephrops trawlers.

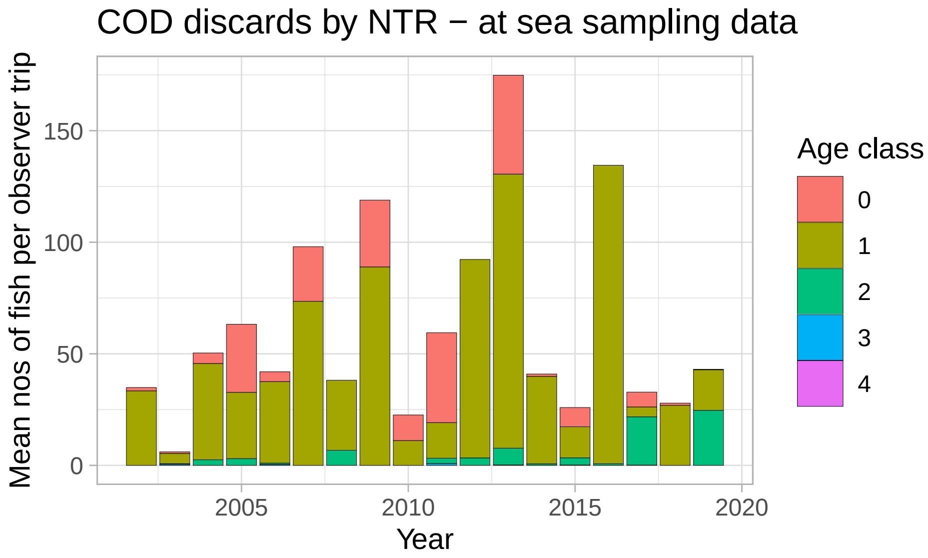

Figure 2.6: Mean numbers of fish at age discarded per observer trip for cod (COD). NTR - Nephrops trawlers.

The survey data is summarized in Figures 2.7 - 2.12, in numbers of fish caught per hour of survey effort. Note that figures are separated for the quarter 1 (Figures 2.7 - 2.9; Q1) and quarter 4 (Figures 2.10 - 2.12; Q4) surveys. The Q1 survey for haddock is composed of mostly age 1 to age 3 fish, and the index increases over the time series, particularly after 2010, and there was a peak value in 2019, substantially higher than any of the previous years. The Q1 survey caught mostly age 1 whiting, and some age 2 fish. The survey abundance index for whiting seems to reach a maximum between late 1990's and early 2000's, after that decreases, with two peaks of age 1 fish in recent years (2014 and 2016). For cod, the survey is able to catch a wider range of age groups compared with haddock and whiting, with age classes from age 1 to age 4. However, the numbers of cod caught are very low, less than 30 fish per hour of trawling, except for a peak of age 1 fish in 1987 that distorts the scale of the time series.

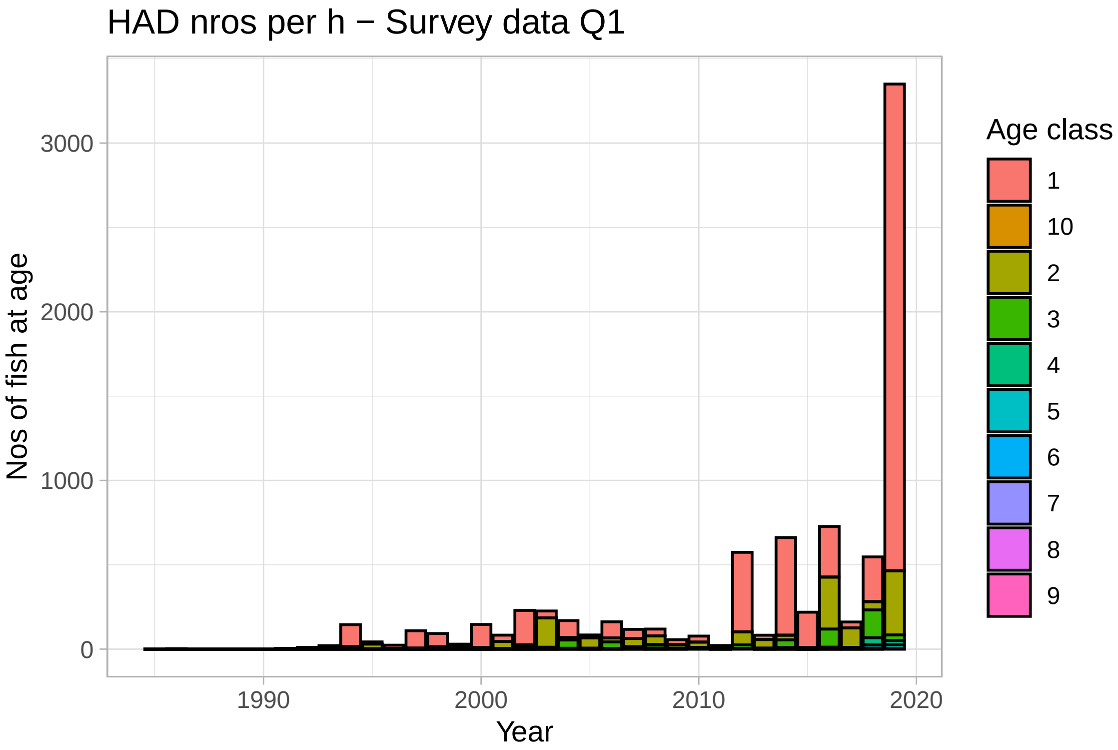

Figure 2.7: Numbers of fish at age caught per hour by the Q1 survey for haddock (HAD).

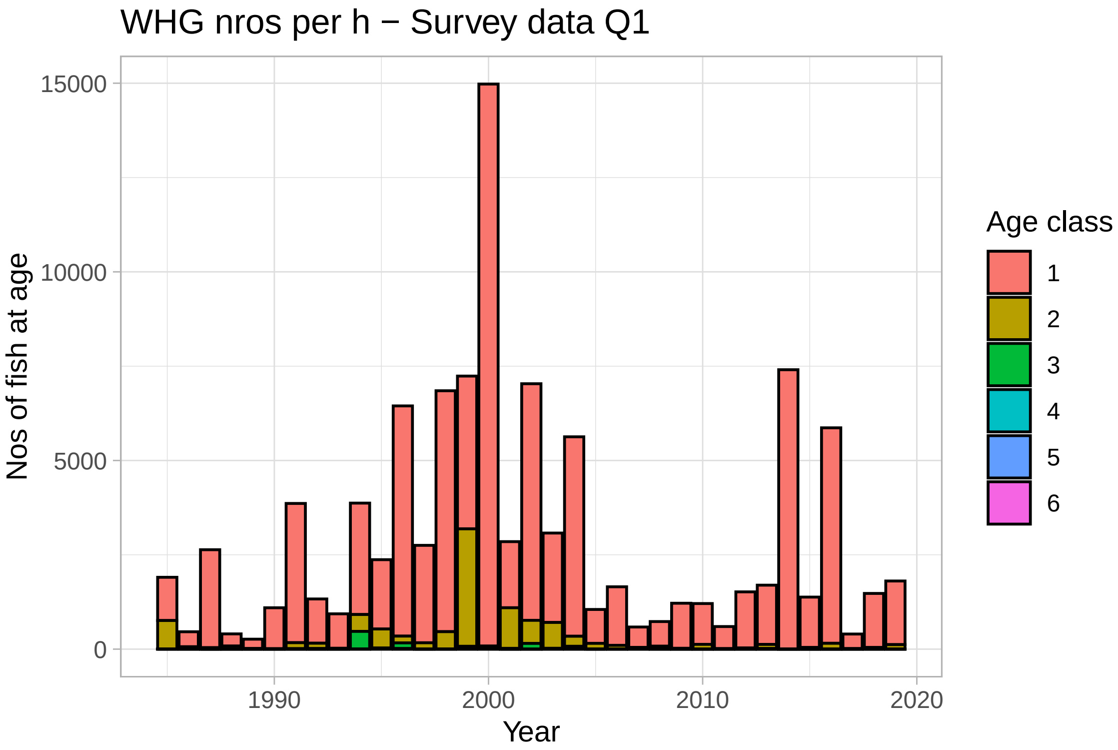

Figure 2.8: Numbers of fish at age caught per hour by the Q1 survey for whiting (WHG).

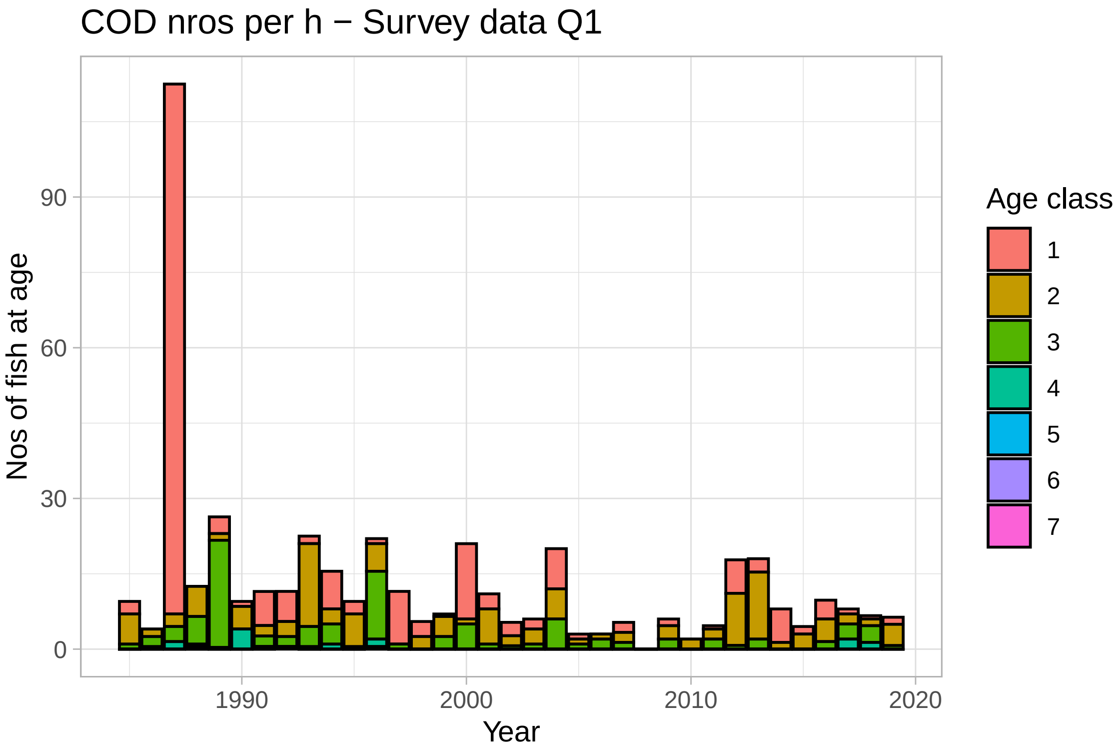

Figure 2.9: Numbers of fish at age caught per hour by the Q1 survey for cod (COD).

The Q4 survey contains more missing data for all three species than the Q1 survey. The Q4 survey for haddock is composed of age 0 fish in the most recent years, and some age 1 fish. Before 2014, the survey abundance for haddock is very low. Records of the Q4 survey for whiting are concentrated after late 1990's. Large amounts of age 0 whiting were caught by the Q4 survey, and some age 1 fish to a lower extent. The Q4 survey for cod contains sparse data. There seems to be older fish caught at the early part of the time series, mostly age 1 to age 3 fish, while for the most recent years the survey caught more young age 0 fish. For all three species, the survey (in particular the Q1 survey) seems to be able to catch more older fish when compared with the Nephrops trawlers. This can be because the survey was designed to sample demersal fish species, with groundgear, wire sweep rig and trawl net all adapted to efficiently catch the widest range of fish species and sizes that live close to the sea bottom. In contrast, Nephrops trawlers do not target whitefish and operate at different depths and in specific areas where Nephrops can be fished (Figures 2.2 and 2.3).

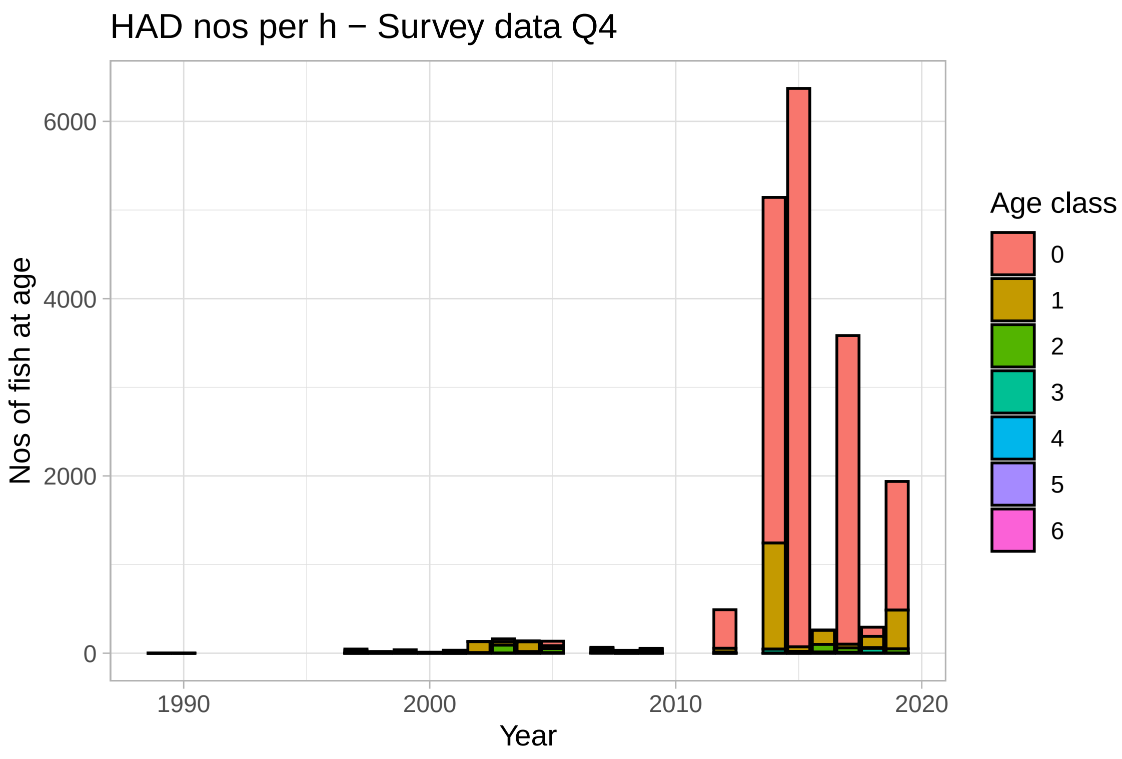

Figure 2.10: Numbers of fish at age caught per hour by the Q4 survey for haddock (HAD).

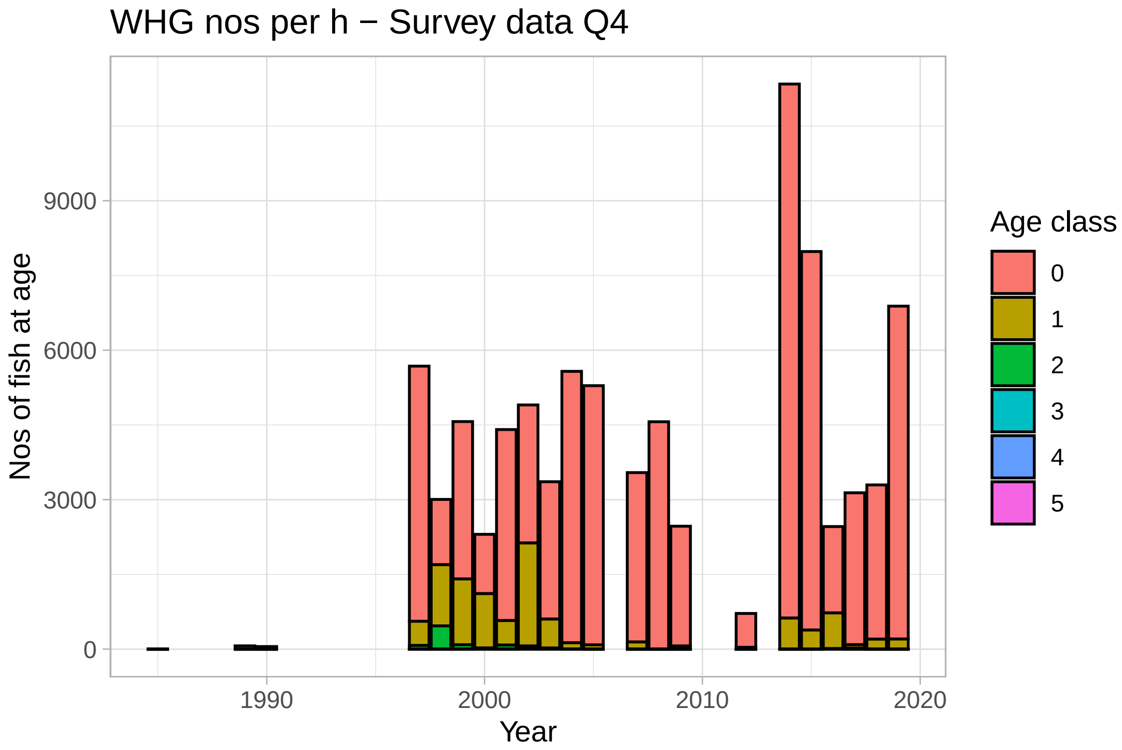

Figure 2.11: Numbers of fish at age caught per hour by the Q4 survey for whiting (WHG).

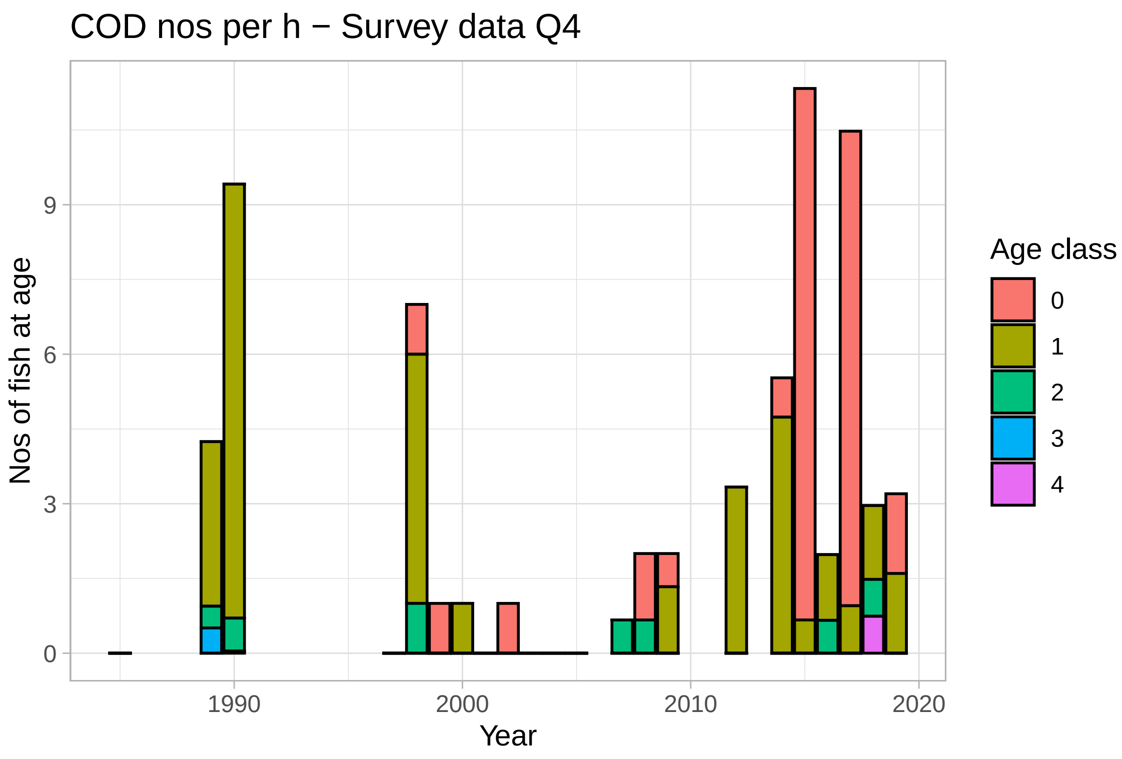

Figure 2.12: Numbers of fish at age caught per hour by the Q4 survey for cod (COD).

The mean weights at age for each species were calculated separately using the discards data, the Q1 survey and the Q4 survey (green, red and purple lines respectively, Figures 2.13 - 2.15). The “raw” mean weights at age have unrealistically high variability from year to year. There are also data gaps for the survey data, particularly at older ages when there are fewer fish.

The Q4 weights are higher than the Q1 weights since fish grow during the year despite being the same age class. The discard mean weights are in agreement with the mean weights from the Q1 survey. One possible explanation is that the length-weight parameters used are from the midpoint of the year, which are closer to the parameters used for the Q1 survey that operates in March. The Q4 survey occurs in November, and the length-weight parameters used are distinct. Mean weight at age 5 for haddock is zero for the discards data because there were no age 5 fish caught by the fishery.

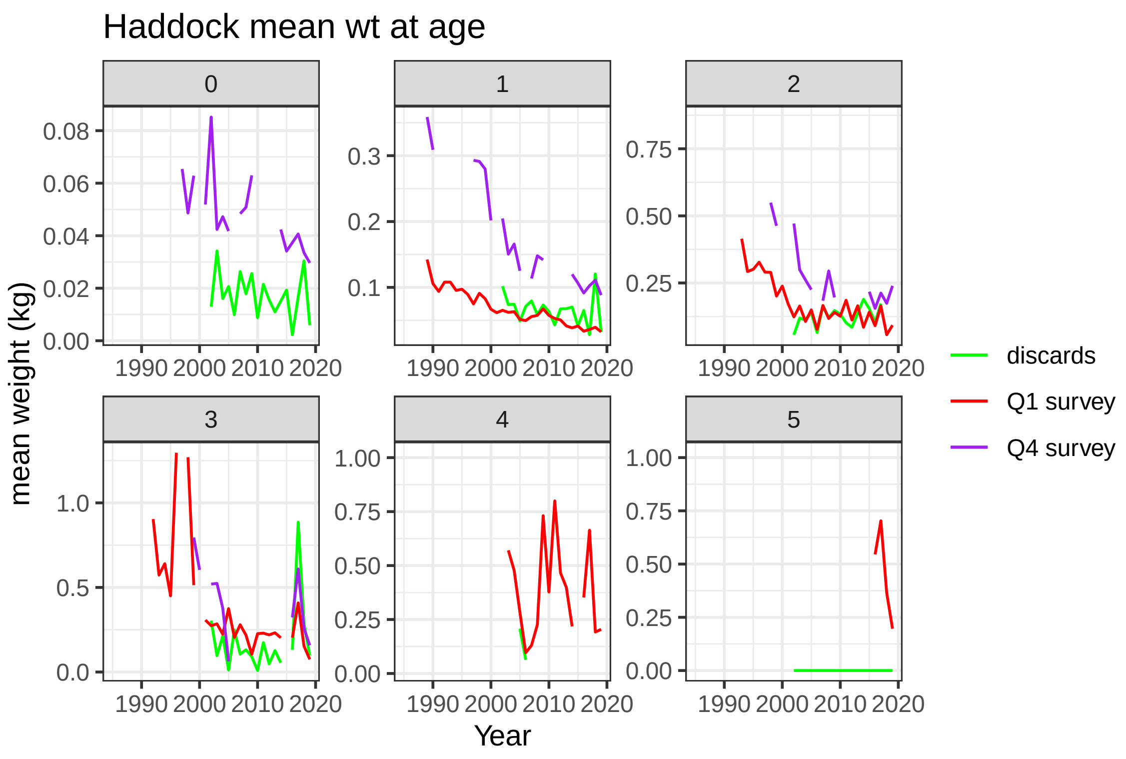

Figure 2.13: Mean weight at age of an individual fish calculated from raw data for haddock.

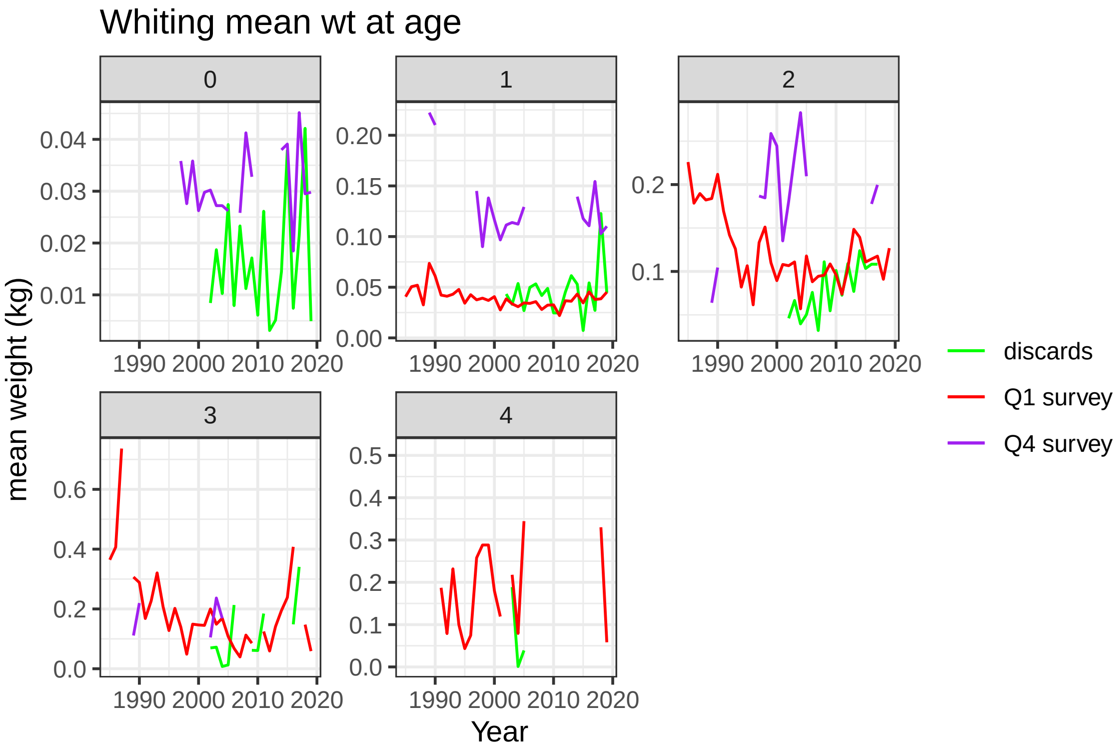

Figure 2.14: Mean weight at age of an individual fish calculated from raw data for whiting.

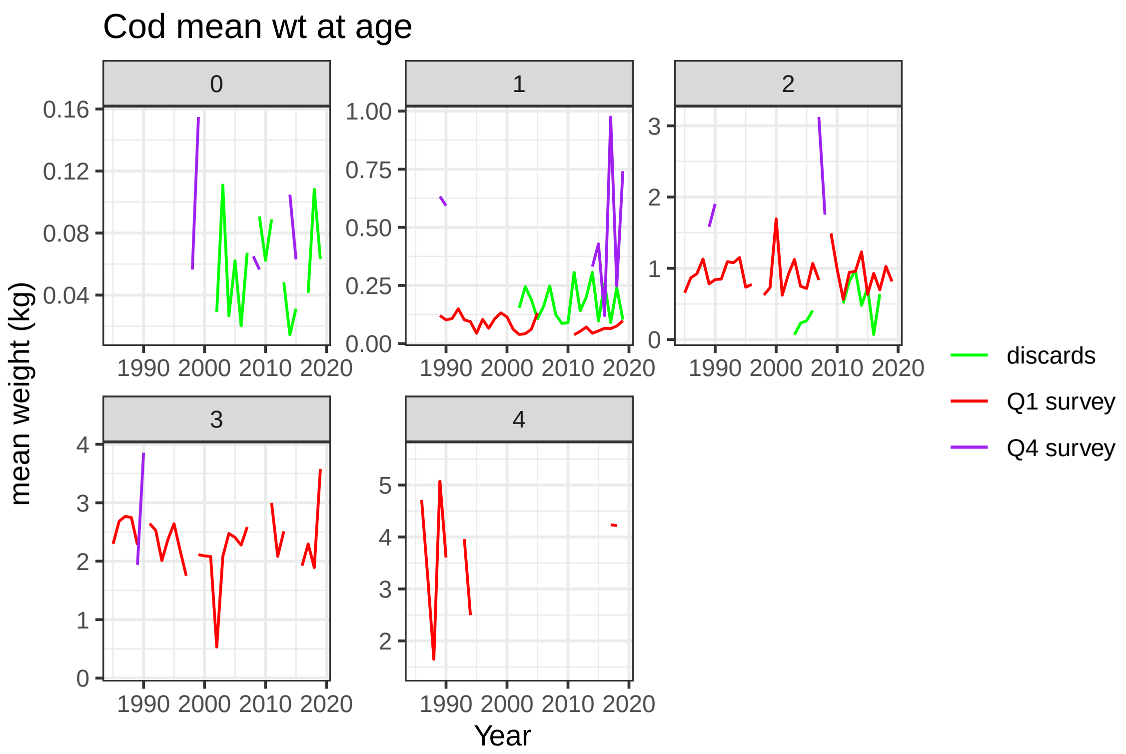

Figure 2.15: Mean weight at age of an individual fish calculated from raw data for cod.

The "smoothed" mean weights at age are shown in Figures 2.16 - 2.18. The "smoothed" weights represent a more realistic assumption on the variability of mean weight at age (reduced noise in the data). The average weight from the Q1 and Q4 surveys is shown as "surv Clyde" (purple line). Two data points had to be excluded from the haddock data set (values for 1996 and 1998 for age 3), otherwise these would create an artificial peak on the mean weights around 1997. When averaging the smooth weights from Q1 and Q4 surveys, the mean weight for haddock between 1985-1990 had to be considered constant over time because there were very few fish caught by the survey before 1990. For haddock, the mean weights at age for the Clyde are larger than the ICES weights, and show a stronger decline over the time series. In contrast, for whiting, the Clyde mean weights are smaller than the ICES weights for the older ages. For cod, there is some degree of overlap between the Clyde mean weights and the ICES weights, and both show a decline in mean weight at age over time, particularly for ages 3 and 4. Since there are very few fish caught older than age 2, these will have a small contribution for the further calculations of spawning stock biomass. Plots of model fit to the data points (and the excluded data points for haddock) for each age class are shown in Appendix A (Figure A.4).

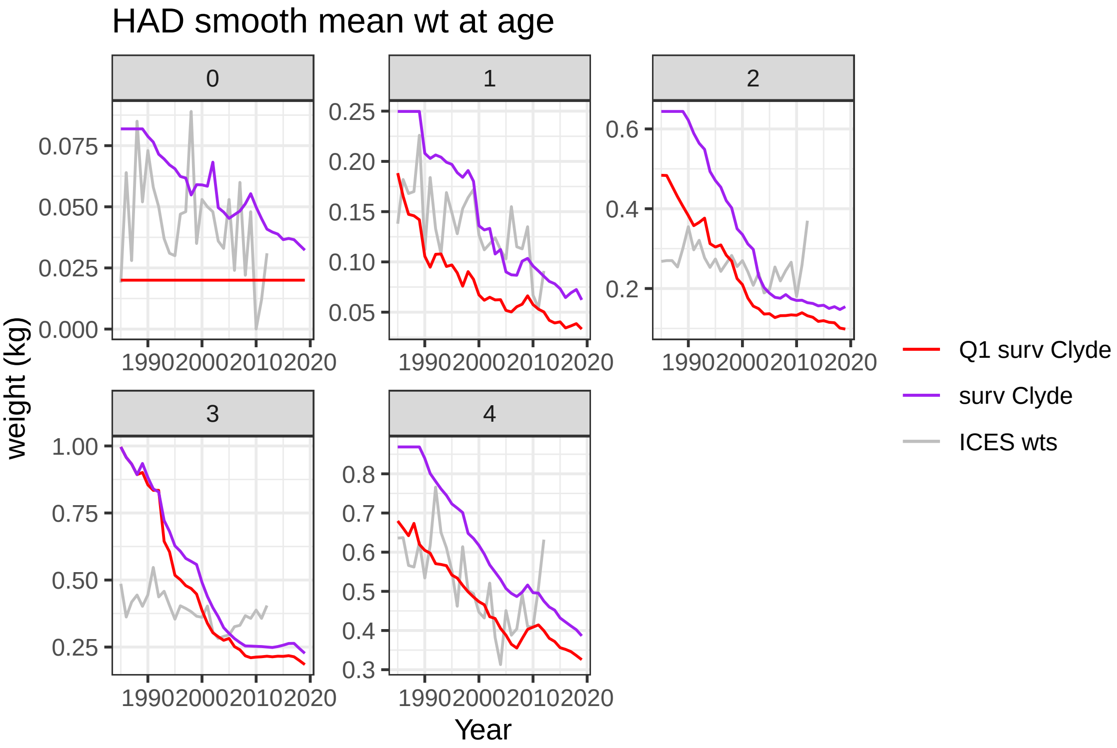

Figure 2.16: Smooth mean weight at age of an individual fish calculated for haddock (HAD).

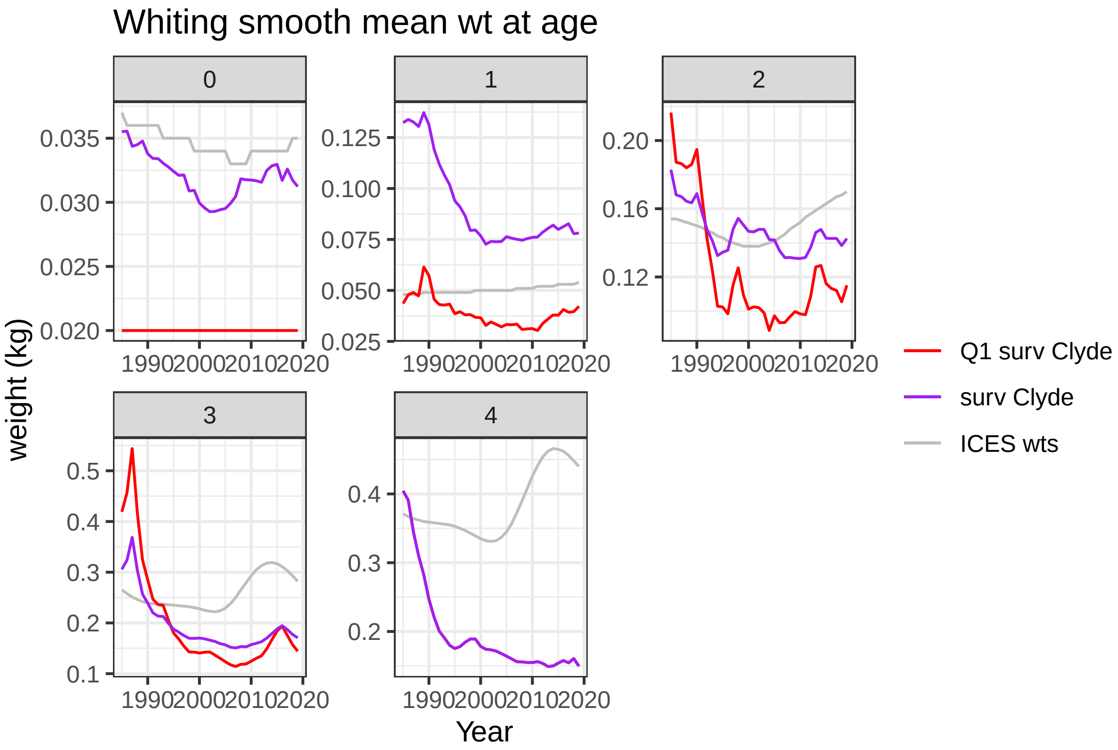

Figure 2.17: Smooth mean weight at age of an individual fish calculated for whiting (WHG).

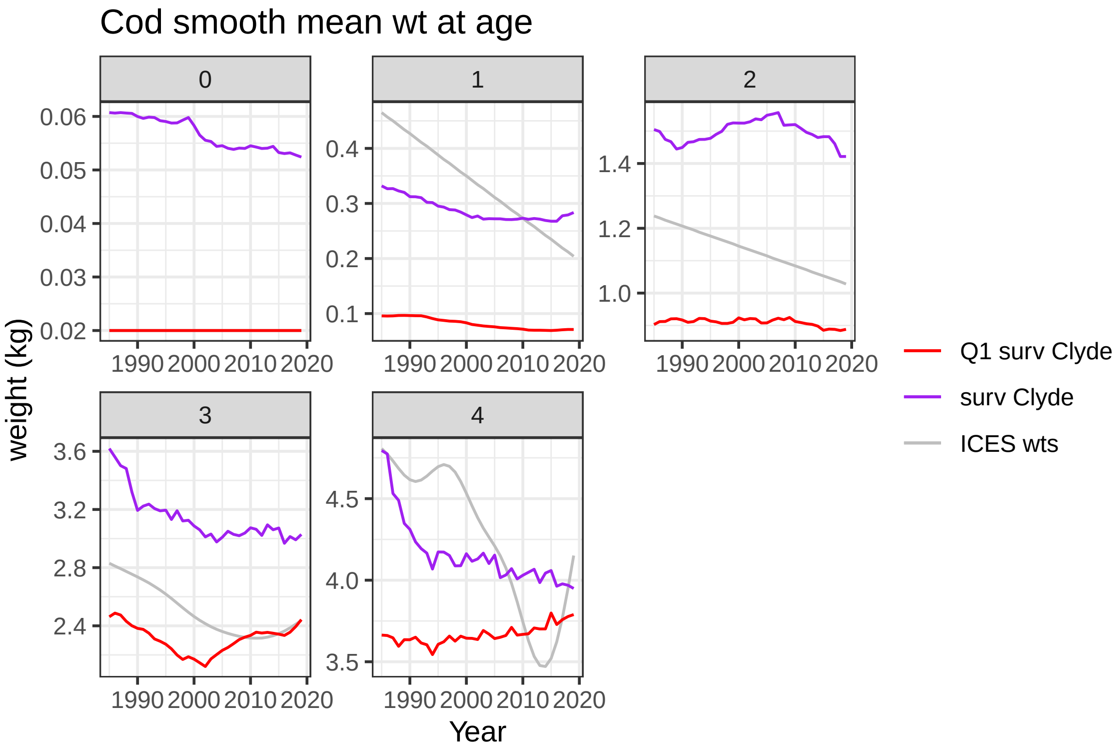

Figure 2.18: Smooth mean weight at age of an individual fish calculated for cod (COD).

There is a large discrepancy between the weights from the Q1 survey and from an average of the Q1 and Q4 surveys, particularly for cod. This is because there are few observations for the Q4 survey for cod, and cod attains larger sizes over the course of a year than whiting and haddock.

For all three species, there seems to be a consistent decreasing trend of the mean weights at age over the time period, and more pronounced for the older age classes. These results are in line with Hunter et al. (2016) that showed there have been substantial decreases in the lengths of most age groups of Clyde haddock and whiting due to declines of growth parameters.

2.4 Conclusions

After data extraction, cleaning and processing, the following data types will be used for the next Chapters (see Figure 1.1 at the end of Chapter 1). The observer data is used as numbers at age and as discard weight per observer trip in all subsequent Chapters. The Nephrops landed weights per observer trip as auxiliary information is used in Chapter 3 only. The logbook data as number of fleet trips and Nephrops landed weights by the commercial fleet is used in Chapter 3 to raise the discard estimates to the fleet level. The survey information in numbers at age or weight caught per hour of survey effort (separately for the Q1 and Q4 surveys) is also used in all following Chapters. The mean weights at age estimated from the survey data are chosen for further analysis since they provide a longer time series and are in close agreement with the discard weights. Both Q1 and an average of Q1 and Q4 survey weights are used in the following chapters, depending on the purpose for which they are being used (e.g. calculating stock biomass, or natural mortality, or estimated catches).

CHAPTER 3 - Estimating discards for the Clyde Nephrops fishery

3.1 Introduction

Commercial fishing with non-selective gears such as bottom trawls captures a large number of other species and sizes alongside the target species which are often rejected due to economic or legislative reasons (bycatch). For a clear definition, bycatch is defined as any organism caught unintentionally, while discards are the portion of the catch which is not used and thrown away at sea. Discards can represent a major proportion of total catch and are generally considered as a source of waste of natural resources. For the North Sea beam trawl and otter trawl fisheries, one third of total weight caught was discarded, of which 60 to 70% represent roundfish and flatfish species (Catchpole et al., 2005). Borges et al. (2005) estimated around 20 000 tonnes of fish discarded every year (1993 - 2002) by the Irish demersal fishery, representing one third of the total catch and mostly species with commercial interest (e.g. haddock, whiting, dab, plaice). In the West of Scotland, the Nephrops trawling fleet discarded 40 to 60% of total catch weight between 1987 and 2009 (Fernandes et al., 2011). Over the last 30 years, trawling for Norway lobster has become the most important fishing activity in the Firth of Clyde, after the collapse of the whitefish fishery in early 2000s (Thurstan and Roberts, 2010). The Clyde Sea represents one of the main grounds for Nephrops trawling, accounting for 24% of total Nephrops landed in Scotland (Mills et al., 2017). Two studies on bycatch and discard composition in the Nephrops fishery specifically conducted in the Clyde estimated high rates of discarding, around 70% of total catch (Bergmann et al., 2002a; Stratoudakis et al., 2001). These discards were comprised mostly of immature whitefish such as haddock and whiting of small sizes. Fish caught below minimum landing sizes or of small sizes but with low economic value are one of the reasons for discarding to occur. Other causes are related to legal restrictions such catch quota limitations (when a vessel reaches the amount of quota allowed for a species) and capacity constraints (when a vessel reaches full storage capacity and only keeps fish with high value; Cook, 2001; Cook, 2019a). The implementation of the Registration of Buyers and Sellers Scheme 2005 (Scottish Statutory Instrument, 2005) in Scotland which strictly regulates the fish auctions at ports meant that illegal fish could not be landed, which might in turn have contributed to an increase of discarding (Fernandes et al., 2011). In the case of the Nephrops fishery in the Clyde, most boats do not hold licenses for selling fish and most of the fish bycatch is undersized. The discards problem is a well-known issue outside the fisheries management and academic field. Public concern raised by a media campaign and supported by environmental non-governmental organizations (NGOs) put pressure on fisheries managers and led to the introduction of the "Landing Obligation" in 2013 (Borges, 2015). This piece of legislation requires all fish that are caught to be landed at port, effectively acting as a discard ban. However, a landing obligation alone without changes in fishing practices might have negative consequences to other species such as seabirds, marine mammals and seabed fauna, which depend on discarded fish as a food source (Heath et al., 2014).

Estimates of discard quantities are needed to properly quantify the impacts of fishing on fish stocks and broader ecosystem effects. The mandatory collection of discard data in European member-states started in 2002 by national sampling programmes (Data Collection Framework; European Commission, 2001) that would feed into international stock assessments routinely done by ICES (International Council for the Exploration of the Sea). These sampling schemes consist of allocating observers aboard individual vessels to collect data on quantity and composition of fish discarded at sea. Usually, observers are placed on a small fraction of vessels and fishing trips, covering less than 1% of the fleet (Stratoudakis et al., 2001). The sampled discard data needs to be raised to represent the whole fishery. The most used methods to estimate discards at the fleet level are based on the sampling design ("design-based" estimators). An ICES review group (ICES, 2007) has summarized the available design-based procedures, namely mean and ratio-based estimators. Defining the fishing fleet as the target population being sampled, the sampled data should be readily raised using a factor available at the population level and assuming that the unobserved portion of the fleet behaves in the same way as the observed part (Cotter and Pilling, 2007). Assuming the primary sampling unit is the fishing trip and a simple random sampling design, the discards for the fishery can be estimated by multiplying a mean from the observed discards trips and the total number of fishing trips in a year. This constitutes the simplest mean estimator.

The ratio estimator uses an auxiliary variable that is proportional to the discard quantities to calculate the population total (Thompson, 2012). Auxiliary variables can be fishing time, landings of all species or a component of the landings (Vigneau, 2006). Several studies have used and compared the sample mean estimator and versions of the ratio estimator with alternative auxiliary variables (Stratoudakis et al., 1999; Stratoudakis et al., 2001; Borges et al., 2005; Millar and Fryer, 2005). Stratoudakis et al. (1999) compared 6 different landing components as auxiliary variables and proposed a collapsed ratio estimator to merge data from separate sampled areas. Raising the discard sampled data using fishing effort in hours as an auxiliary variable in Borges et al. (2005) gave significantly higher discard estimates when compared with other variables (number of trips, total landings of all species). The confidence intervals calculated for the design-based estimators based on the classical mean plus standard error can be unrealistically wide, so a complementary approach based on bootstrap methods can improve uncertainty estimates of the design-based approaches (Catchpole et al., 2011). On the other hand, model-based estimators can also be applied for estimating discards at the fleet level, with the flexibility of including other sources of data such as scientific trawl surveys. Heath and Cook (2015) proposed a model that splits the discarding process into two components – size and bulk discarding. The size-related discarding occurs due to legal requirements or low commercial value of small individuals, while the bulk proportion represents fish that are rejected regardless of size, and it may occur when a vessel reaches its quota. Additional length data from scientific trawl surveys was used to inform the size-related discard proportion. A Bayesian inference model was shown to provide more reliable estimates of historical bycatch compared to design-based methods (Breivik et al., 2017). Model-based approaches also allow exploration of the spatial and temporal variability of discarding patterns (Bjørge et al., 2013; Baerum et al., 2019; Feekings et al., 2012).

In this chapter, observer data on discards of demersal fish species by the trawl fishery targeting Norway lobster (*Nephrops norvegicus*) in the Firth of Clyde is analysed. In this fishery, nearly 100% of fish bycatch is assumed to be discarded. The mean and ratio estimators are used to calculate discards at the fleet level and compared to modelled versions of the design-based methods. Model-based approaches can provide better estimates of uncertainty and allow the inclusion of other sources of data, so we explored alternative models that integrate scientific trawl survey data. These different statistical approaches are crucial for accurate estimation of discards of relevant commercial species and non-target species, and for integration of these estimates in fish stock assessment models.

3.2 Methods

3.2.1 Data

The observer data used were the fish discards weight (for cod, haddock and whiting) and the Nephrops landed weight for each observer trip, in the Clyde between 2002 and 2019. The logbook data comprise the total number of fishing trips and Nephrops landings weight for the Nephrops fishery in the Clyde, from 2002 to 2019. The survey data consists of 4 surveys (SWC-IBTS Q1, SWC-IBTS Q4, SCOWCGFS Q1 and SCOWCGFS Q4), in weight of fish per hour of trawling, covering the year range between 1985 and 2019, but only data from 2002 onwards was used to match the same years of the observer and logbook data (see section 2.2.4 in Chapter 2).

3.2.2 Estimation of total annual fish discards

The total biomass of cod, haddock and whiting discarded annually by the Clyde Nephrops fishery was calculated using three different approaches: (1) a mean estimator, (2) a ratio estimator and (3) a combined modelling approach using the survey data. The analysis was conducted for each species on an annual basis due to the scarce number of sampled trips for some years. Data was pooled for all statistical rectangles that include the Firth of Clyde (39E4, 39E5, 40E4, 40E5; Figure 2.1 in Chapter 2). The mean estimator (1) calculates the mean discard weight per trip and then uses the total number of fleet trips to raise the discards to fleet level. The ratio estimator (2) assumes a linear relationship between discards and Nephrops landings, and allows the calculation of a slope that measures how fast discards per trip increase with landings per trip. For the ratio estimator, discards are raised using Nephrops landings at the fleet level. The mean and ratio estimators provide independent point estimates of discards each year. It is likely, however, that discards in successive years are correlated. A random walk model was explored that assumes that the estimate in year y is a predictor of the estimate in year y+1. This component acts a time series smoother, removing the noise in the data and helping to identify the main trends. The mean and ratio estimators can be calculated using analytical methods (design-based estimators). For inclusion of the time series smoother, modelling approaches had to be sought. Models for the simple mean and ratio estimators were also constructed so the parameter estimates and credible intervals could be directly compared to the models with a time series smoother. The third approach was a combined model (3), that uses both the Nephrops landings a measure of fishing effort and the survey data as a measure of fish abundance in the sea to estimate discards at the fleet level. Model parameters were estimated using Bayesian methods. The following section describes the equations for each model used to calculate discards for each fish species.

1. Mean Estimator with random walk We assume that the annual quantity of discards, D_y, is a function of the mean discards per trip, $\bar{d}_y$, and the total number of trips T_y:

$$D_y = \bar{d}_y T_y \tag{3.1}$$

Here, $\bar{d}_y$ is a parameter to be estimated, while T_y is a known quantity. The design-based mean is simply calculated by summing discards across observed trips (per year) and dividing by the number of observer trips (per year). Equations for the design-based mean, variance and 95% confidence intervals are shown in Appendix B (Supplementary Information). Note that the design-based mean estimator does not include a random walk. It is likely that discarding in successive years is correlated as there will be similarities in the fishery and stocks from year to year. Hence we may assume that $\bar{d}_y$ follows a random walk on a log scale to exploit any correlation between years and reduce the effective number of parameters to be estimated:

$$\bar{d}_y = \bar{d}_{y-1} exp(\varepsilon_y) \tag{3.2}$$

where $\varepsilon_y \sim normal(0, \sigma_r)$. There are multiple observed values of discards per trip, $\hat{d}_{t,y}$, each year and these are assumed to have lognormal errors so that:

$$\hat{d}_{t,y} \sim lognormal(log(\bar{d}_y), \sigma_d) \tag{3.3}$$

where σ_d is the standard deviation of the observation error distribution. If an estimate of $\bar{d}_y$, is $\bar{\bar{d}}_y$, the estimated total discards, $\hat{D}_y$ can then be calculated from the total number of trips:

$$\hat{D}_y = \bar{\bar{d}}_y T_y \tag{3.4}$$

Note that if we omit the random walk from the model, it reduces to a model very similar to the design-based mean estimator but with a lognormal error assumption and constant variance across years.

2. Ratio Estimator with random walk The ratio estimator relies on the assumption of a linear relationship between fish discard weight per trip (d_t) and an auxiliary variable that accounts for a measure of fishing effort such as fishing hours or landed biomass. As previously explained in Chapter 2, the Nephrops landings per trip can be used as a proxy for fishing effort. We then assume that the discards per trip, $d_{t,y}$, are related to the Nephrops landings per trip, $l_{t,y}$, by a common annual ratio, r_y:

$$d_{t,y} = r_y \; l_{t,y} \tag{3.5}$$

Here, r_y is a parameter to be estimated, while $l_{t,y}$ is a known quantity.

A yearly ratio is calculated directly by summing discards across observer trips divided by landings across observer trips. Then total discards per year can be calculated by multiplying the yearly ratio by the Nephrops landings by the fleet. Equations for the design-based ratio, variance and 95% confidence intervals are shown in Appendix B (Supplementary Information). Note that the design-based ratio estimator does not include a random walk.

Similar to the mean estimator approach we may assume that r_y follows a random walk to exploit any correlation between years and reduce the effective number of parameters to be estimated:

$$r_y = r_{y-1} exp(\varepsilon_y) \tag{3.6}$$

where $\varepsilon_y \sim normal(0, \sigma_r)$. The observed values of discards per trip, $\hat{d}_{t,y}$ are assumed to have lognormal errors so that:

$$\hat{d}_{t,y} \sim lognormal(log(d_{t,y}), \sigma_d) \tag{3.7}$$

where σ_r is the standard deviation of the observation error distribution. If an estimate of the ratio r_y is $\hat{r}_y$, the estimated total discards can then be calculated from the total Nephrops landings by the fleet, L_y:

$$\hat{D}_y = \hat{r}_y L_y \tag{3.8}$$

Note that if we omit the random walk from the model, it reduces to a model very similar to the design-based ratio estimator but with a lognormal error assumption and constant variance across years.

3. Combined model The mean and ratio estimators do not explicitly account for the biomass of fish in the sea. It might be expected that the amount of discarding is related to the biomass of fish where higher abundance leads to more discards. The combined model developed here is an extension of the ratio estimator that uses additional information from the survey data. Assuming that average fish discards per year ($\bar{d}_y$) is proportional to the biomass of fish in the sea (B):

$$\bar{d}_y \propto \; B_y \tag{3.9}$$

Converting into equation with a proportionality constant, Q:

$$\bar{d}_y = Q \; B_y \tag{3.10}$$

Here, Q is a proportionality constant, B_y is a parameter to be estimated.

Although the biomass, B, is an unknown parameter there are relative abundance indices calculated from the research survey data. It is possible therefore to estimate B subject to constraints in a similar fashion to factor analysis (e.g. Conn, 2010; Cook et al., 2021). If we assume the kth survey biomass index, $u_{k,y}$, is proportional to B_y with a catchability coefficient q_k we have:

$$u_{k,y} = q_k B_y \tag{3.11}$$

with observations $\hat{u}$ of u from multiple surveys and assuming lognormal errors:

$$\hat{u}_{k,y} \sim lognormal(log(u_{k,y}), \sigma_k) \tag{3.12}$$

Provided one of the q_k values is fixed (e.g. $q_1 = 1$ implying $u_{1,y} = B_y$) the remaining values of q_k can be estimated and will give estimates of B on a relative scale. For example, if $q_1 = 1$ then the biomass index B will be on the scale of survey index 1. Some of the fish biomass in year y will be survivors from year $y - 1$ so the biomass is modelled as a random walk:

$$B_y \sim lognormal(log(B_{y-1}), \sigma_B) \tag{3.13}$$

In addition to the survey indices, we have observations of the discards per trip and making the same error assumption leads to:

$$\hat{d}_{t,y} \sim lognormal(log(\bar{d}_y), \sigma_d) \tag{3.14}$$

These observations allow the estimation of Q and B_y, hence the total quantities of fish discarded each year can be calculated from the number of trips as:

$$\hat{D}_y = \hat{Q}\hat{B}_y T_y \tag{3.15}$$

where the hatted quantities are estimates of the values after fitting the model. In this combined model, any correlation in the annual discard rates is explained by correlation in the biomass time series rather than the discard rate parameters as used in previous models.

An extension of the model allows the relationship to be non-linear and take the form:

$$\bar{d}_y = QB_y^{\beta} \tag{3.16}$$

This model can be fitted and total discards calculated in a similar way to the linear version. A further version of this model is to account for fishing effort as Nephrops landings (see section 2.2.2 in Chapter 2):

$$d_{t,y} = QB_y l_{t,y} \tag{3.17}$$

Here, Q differs from the proportionality constant Q in equation 3.10 (which reflects the mean estimator) and integrates the ratio estimator, r_y, as in earlier models. This means that the longer the vessel fishes for and the more fish are in the sea, the higher the quantities of discards. The observations of discards per trip make the same error assumption that leads to:

$$\hat{d}_{t,y} \sim lognormal(log(d_{t,y}), \sigma_d) \tag{3.18}$$

These observations, along with the survey index observations, allow the estimation of Q and B_y. The total quantities of fish discarded each year can be calculated from the Nephrops landings:

$$\hat{D}_y = \hat{Q}\hat{B}_y L_y \tag{3.19}$$

where the hatted quantities are estimates of the values after fitting the model. An extension of the model is to allow the relationships with biomass and landings to be non-linear and take the form:

$$d_{t,y} = QB_y^{\beta_1} l_{t,y}^{\beta_2} \tag{3.20}$$

This model can be fitted and total discards calculated a similar way to the linear version but requires the estimation of two additional parameters.

3.2.3 Model configuration and parameter estimation

Priors on the parameters were chosen to be uniform or log uniform (Table 3.1). The prior distribution on $\bar{d}_y$ was assumed to be log uniform. This will be a weakly informative prior but avoids negative values.

Table 3.1 - Model configurations and prior distributions on the parameters. σ_r represents the error term for the random walk of the mean/ratio estimator; σ_b the error term for the random walk of biomass; σ_d is the observation error on the discards data, and a uniform prior was used in all models ($\sigma_d \sim uniform(0, 10)$). exp represents exponents.

Model	Estimator	Time series	Priors
1.0 Mean	Mean	No	$log(\bar{d_y}) \sim uniform(-10, 10)$
1.1 Mean	Mean	Yes	$log(d_1) \sim uniform(-10, 10)$
			$\sigma_r \sim uniform(0, 10)$
2.0 Ratio	Ratio	No	$log(r_y) \sim uniform(-10, 10)$
2.1 Ratio	Ratio	Yes	$log(r_1) \sim uniform(-10, 10)$
			$\sigma_r \sim uniform(0, 10)$
			(all priors for combined models)
3.0 Comb model	Mean	Survey biomass	$log(B_1) \sim uniform(-5, log(max(u_{1,y})))$
3.1 Comb model	Ratio		$q \sim uniform(-10, 10)$
3.2 Comb model exp	Mean		$\sigma_B \sim uniform(0, 10)$
3.3 Comb model exp	Ratio		$\sigma_k \sim uniform(0, 10)$
			$Q \sim uniform(-100, 100)$
			$\beta \sim uniform(0, 2)$

All models were set up in “RStan” (Stan Development Team, 2023) and applied separately to each species (cod, haddock and whiting). For the combined model, each survey index was standardized to the series mean for convenience (see further explanation in Appendix B). Preliminary runs with 3 chains and 10 000 iterations were assessed with trace plots and calculation of the R-hat statistic, which provides a way to inspect mixing across chains and model convergence. Final runs consisted of three MCMC chains with 500 000 iterations, a burn in of 250 000 and a thinning rate of 1000. Standard statistics recorded after each model run were the mean, median and 95% credible intervals for all critical parameters. Histograms of the posterior parameters allowed the shape of the distributions to be inspected to see if there were skewed distributions. The median values were chosen instead of the mean for further analysis as these are less sensitive to asymmetric distributions.

For model comparisons, the deviance information criterion (DIC), effective number of model parameters (pD) and R-squared value were calculated from a log likelihood function based on 750 random samples from the MCMC chains (Spiegelhalter et al., 2002). Equations on how to calculate these are available in Supplementary Information (Appendix B).

3.3 Results

3.3.1 Comparison of design-based and model-based methods

The mean and ratio estimators using design-based methods are compared to mean and ratio estimators computed using Bayesian statistical models (Figure 3.1). Both give similar values of discards in terms of trends and scale. The model-based estimates are slightly lower and give narrower confidence intervals for the later part of the time series (2010 - 2019). The largest discrepancy between the two approaches was for cod, where the estimates of discards from 2012 onward from the models are lower, but still within the range of the 95% credible intervals of the design-based methods. These differences can be partly as a result of using priors, the use of the median rather than the mean and the lognormal assumption for the error distribution in the Bayesian model. The model-based mean and ratio estimators were set up as Bayesian equivalents of the design-based estimators so comparisons could be made using the same approach (either Bayesian or frequentist approach). The model mean and model ratio are used for further comparisons with other modelling approaches in the next section of the results.

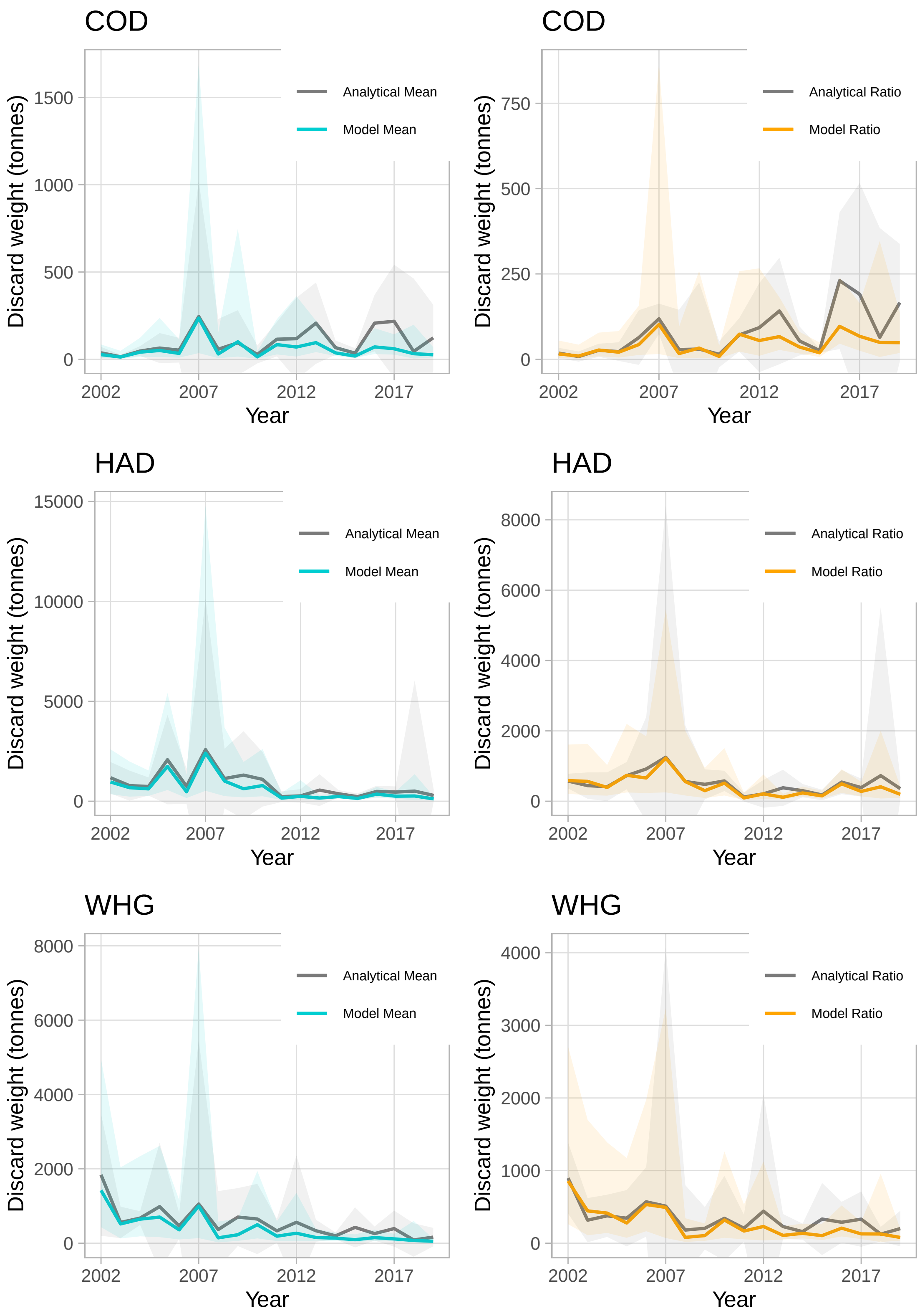

Figure 3.1: Discard estimates calculated at fleet level using design-based (analytical) methods (mean and ratio) and model-based methods (mean and ratio) for haddock (HAD), whiting (WHG) and cod (COD). Shaded areas represent 0.95 credible intervals.

3.3.2 Model-based methods

Model outputs and measures of model comparison (DIC, pD and R-squared of geometric mean) for the three species are summarized in Tables 3.2 - 3.4 and Figures 3.2 - 3.4. Values of other parameters are also provided (σ_r, σ_b and β). The common annual abundance trend (B_y) and the survey data plotted on log scale used for the combined models for each species are available as Figure B.2 in Appendix B.

The error disturbance term (σ_r) for the random walk gives a measure of the variability of the time series of mean discards/discard ratio. For higher values of σ_r, there is less correlation in the data since it means increased variability of the parameter being modelled and smaller smoothing effect. The σ_b follows the same rationale, but for the time series of biomass (B_y) estimated using the survey data. The beta parameters (β_1 and β_2) represent the exponents of the combined model (equations 3.16 and 3.20). It is expected that the beta parameters will be close to 1, if the relationships are linear, meaning that the discards per trip will vary in the same proportion as the landings as a measure of fishing effort and survey data as a measure of the biomass in the sea. If the beta parameters differ significantly from 1, then the relationships are non-linear.

Table 3.2 – Model outputs comparison for models applied to haddock data. Median values for the β exponents are shown together with the 95% credible interval bounds. σ corresponds to σ_r for models 1.0 - 2.1, and to σ_b for models 3.0 - 3.3. pD are the effective number of parameters. ts represents time series component. exp represents exponents.

Model	DIC	pD	R^2	Estimator	σ	β
1.0 Mean	1193.88	18.61	0.99	Mean	NA	
1.1 Mean+ts	1181.84	7.68	0.43	Mean	0.332	
2.0 Ratio	1186.33	19.11	0.99	Ratio	NA	
2.1 Ratio+ts	1171.53	8.41	0.80	Ratio	0.365	
3.0 Comb model	1239.31	13.90	0.29	Mean	0.284	
3.1 Comb model	1220.41	13.98	0.68	Ratio	0.348	
3.2 Comb model exp	1239.74	13.83	0.30	Mean	0.252	$\beta = 1.140$ (0.508-1.931)
3.3 Comb model exp	1219.27	15.57	0.58	Ratio	0.288	$\beta_1 = 0.646$ (0.357-0.938) $\beta_2 = 1.187$ (0.917-1.928)

The mean (1.0) and the ratio (2.0) models have an R-squared of almost 1. This is expected since they are model versions of the design-based estimators and will have a perfect fit to the mean of observations each year (Figure 3.2, grey lines). The dots in Figure 3.2 represent the geometric mean calculated from the data (i.e. mean discard weight per trip), and the lines are the geometric means calculated from the model fit to the data. When including a time series component for the mean and ratio estimators, the DIC and the number of parameters pD decrease substantially, and the models fit less well to the data (R-squared decrease of more than 50%; models 1.1 and 2.1 in Table 3.2).

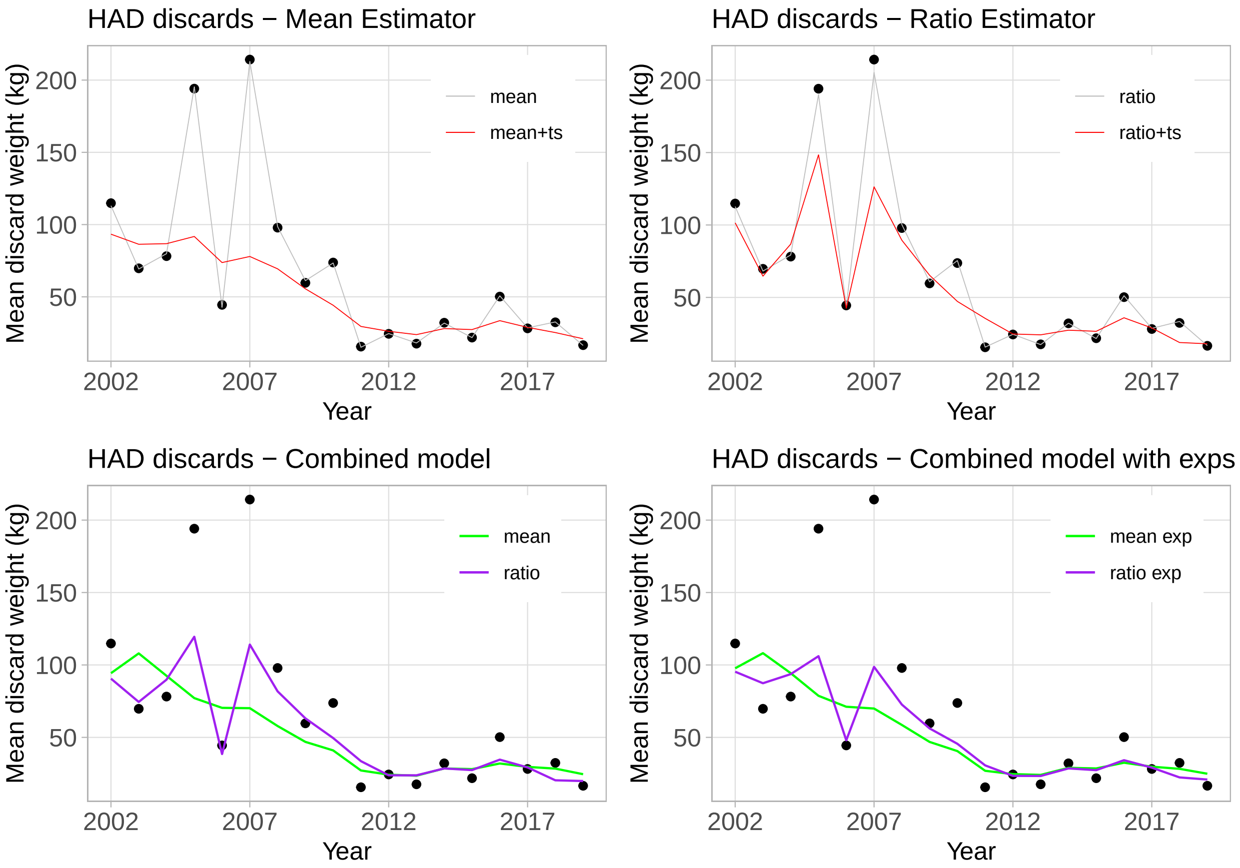

Figure 3.2: Mean discard weight per trip calculated for haddock data for each model. Dots represent geometric mean calculated directly from the data. ts represents time series, exp represents exponents.

The DIC cannot be used as a measure of comparison between the mean/ratio models and the combined model since the latter uses additional data (survey data), which will change the likelihood and hence affect the DIC value. The R-squared calculated from the geometric mean of the data provides information for model comparison. For the different versions of the combined model, the model 3.1 gives one of the lowest DIC with the highest R-squared value. It might be expected that the inclusion of the abundance index from the survey data results in a higher R-squared than the models that do not include the survey data. However, the R-squared value for model 3.1 is still lower than model 2.1 (ratio+ts), giving an indication that the combined models do not outperform the simpler mean/ratio models, and that including the abundance information does not improve the model. The trends in mean discard weight show very similar patterns between all models that include a time series component (Figure 3.2).

Similar outcomes were observed for whiting data (Table 3.3 and Figure 3.3). There is a large decrease in DIC for the mean/ratio models when including a time series smoothing component (20 units decrease in DIC for 1.1 mean+ts). At the same time, R-squared has decreased, meaning the time series models smooth the data, but not to the same extent as for the haddock data. Between the combined models, the model 3.3 (based on ratio and with two exponential terms) gives the lowest DIC and highest R-squared. However, the R-squared value is only marginally better than model 2.1 (ratio+ts) suggesting that, as with haddock, including the abundance data does little to improve the model.

Table 3.3 – Model outputs comparison for models applied to whiting data. Median values for the β exponents are shown together with the 95% credible interval bounds. σ corresponds to σ_r for models 1.0 - 2.1, and to σ_b for models 3.0 - 3.3. pD are the effective number of parameters. ts represents time series component. exp represents exponents.

Model	DIC	pD	R^2	Estimator	σ	β
1.0 Mean	1142.92	19.24	0.99	Mean	NA	
1.1 Mean+ts	1122.55	6.83	0.63	Mean	0.323	
2.0 Ratio	1137.84	18.64	0.99	Ratio	NA	
2.1 Ratio+ts	1119.21	7.20	0.72	Ratio	0.336	
3.0 Comb model	1175.94	11.82	0.66	Mean	0.319	
3.1 Comb model	1173.19	11.86	0.71	Ratio	0.299	
3.2 Comb model exp	1176.74	12.17	0.70	Mean	0.263	$\beta = 1.285$ (0.586-1.940)
3.3 Comb model exp	1168.43	13.75	0.74	Ratio	0.256	$\beta_1 = 0.563$ (0.227-0.860) $\beta_2 = 1.299$ (0.572-1.956)

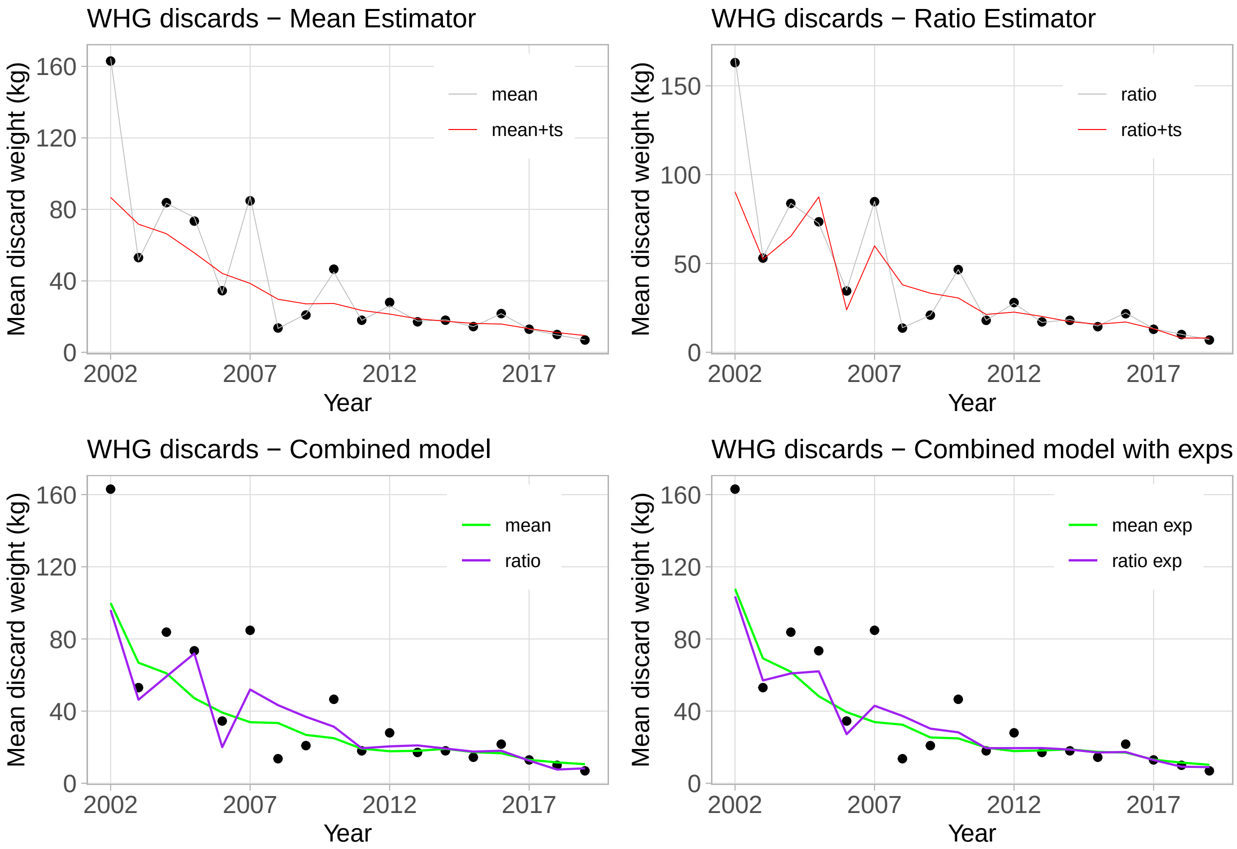

Figure 3.3: Mean discard weight per trip calculated for whiting data for each model. Dots represent geometric mean calculated directly from the data. ts represents time series, exp represents exponents.

Results of the models with a time series component applied to cod data show a poor fit to the data (Table 3.4 and Figure 3.4), particularly for the models based on the mean estimator (R-squared less than 0.2). At the same time, these are the models with lowest DIC and pD. The ratio estimator has higher DIC and incorporates more year-to-year variability of the discards thus having a considerably better fit to the data. The best combined model would be model 3.3 based on the ratio with exponential terms.

Table 3.4 – Model outputs comparison for models applied to cod data. Median values for the β exponents are shown together with the 95% credible interval bounds. σ corresponds to σ_r for models 1.0 - 2.1, and to σ_b for models 3.0 - 3.3. pD are the effective number of parameters. ts represents time series component. exp represents exponents.

Model	DIC	pD	R^2	Estimator	σ	β
1.0 Mean	730.46	18.84	0.99	Mean	NA	
1.1 Mean+ts	718.73	4.97	0.03	Mean	0.172	
2.0 Ratio	739.12	18.85	0.99	Ratio	NA	
2.1 Ratio+ts	724.31	5.76	0.34	Ratio	0.237	
3.0 Comb model	767.12	12.40	0.14	Mean	0.411	
3.1 Comb model	775.51	13.40	0.47	Ratio	0.433	
3.2 Comb model exp	768.12	11.71	0.11	Mean	0.419	$\beta = 0.762$ (0.057-1.814)
3.3 Comb model exp	763.96	12.61	0.27	Ratio	0.417	$\beta_1 = 0.404$ (0.078-0.723) $\beta_2 = 0.771$ (0.090-1.851)

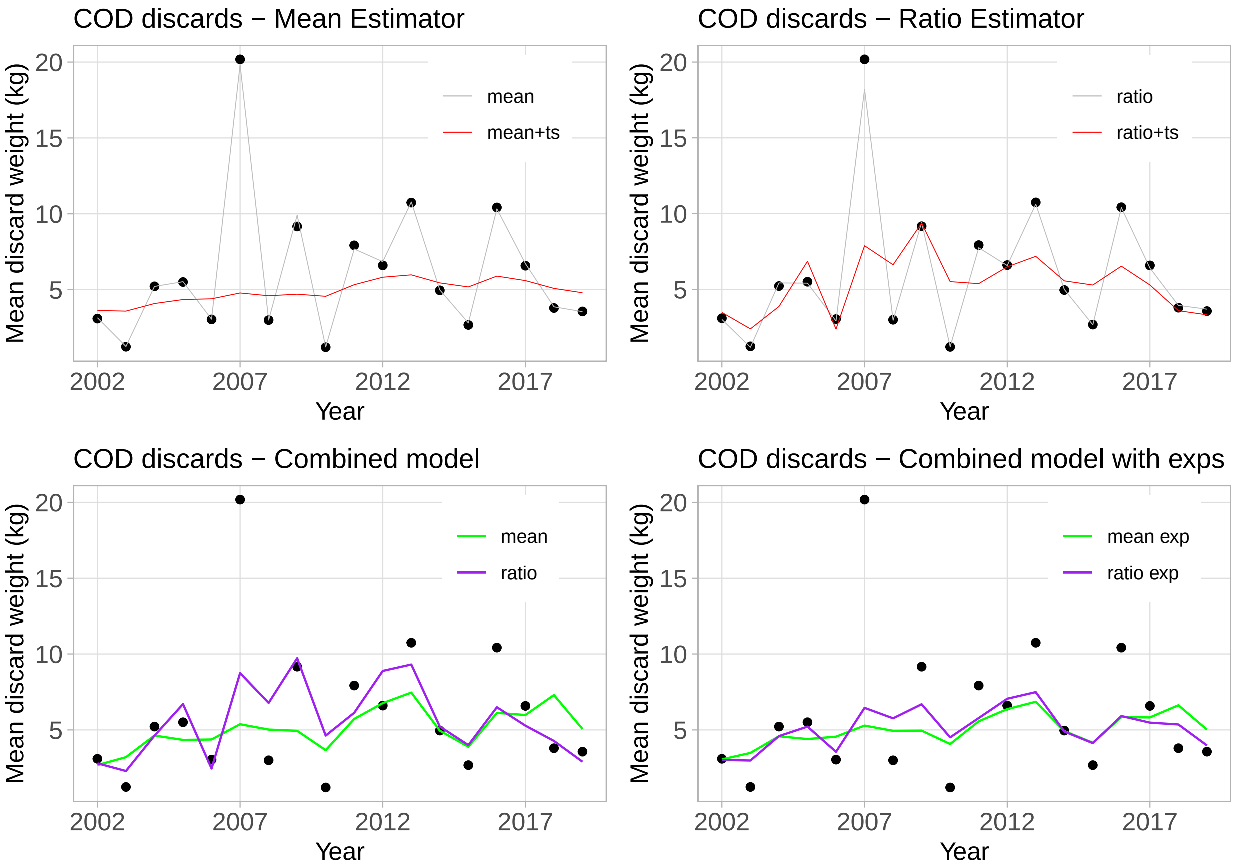

Figure 3.4: Mean discard weight per trip calculated for cod data for each model. Dots represent geometric mean calculated directly from the data. ts represents time series, exp represents exponents.

The lowest σ_r value was estimated for cod, meaning that the time series smoothing had the strongest effect on the mean discards from year to year, which implies most of the annual variability is attributed to observation error. On the other hand, the combined models for cod presented higher values of σ_b of around 0.4 compared with 0.3 for whiting and haddock, which implies that survey abundance temporal trend is more variable for cod.

The beta exponents calculated for the combined models for cod were the lowest (and lower than 1), which implies that the mean discards of cod per trip increase at a much lower rate than increments of both Nephrops landings and survey biomass. For all three species, the values of β_1 were always lower than β_2; so increases in fish abundance in the sea (given by the survey) are translated into slower increases of discards per trip than increases in fishing effort (given by the Nephrops landings).

3.3.3 Comparison of model-based discard estimates at fleet level

Using the results of the model fits described above, estimates of total discards at fleet level can be calculated using the appropriate raising factors (e.g. number of trips or Nephrops landings). In terms of discard weight estimated for the fishery, the comparisons between models are similar across the three species (Figures 3.5 - 3.7). The time series component reduced the uncertainty around the discard estimates giving narrower 95% credible intervals when compared with the simple mean and ratio estimators. The estimates provided by the combined model which include more data/information are not distinct from the ones based on the simpler models. The mean and the ratio estimator (with a random walk term) also give similar discard patterns over time, although the mean estimates are slightly higher than ratio estimates (panel plot “fleet discards - time series”). The results for the combined models 3.2 and 3.3 are omitted, since they are similar to the combined models 3.0 and 3.1.

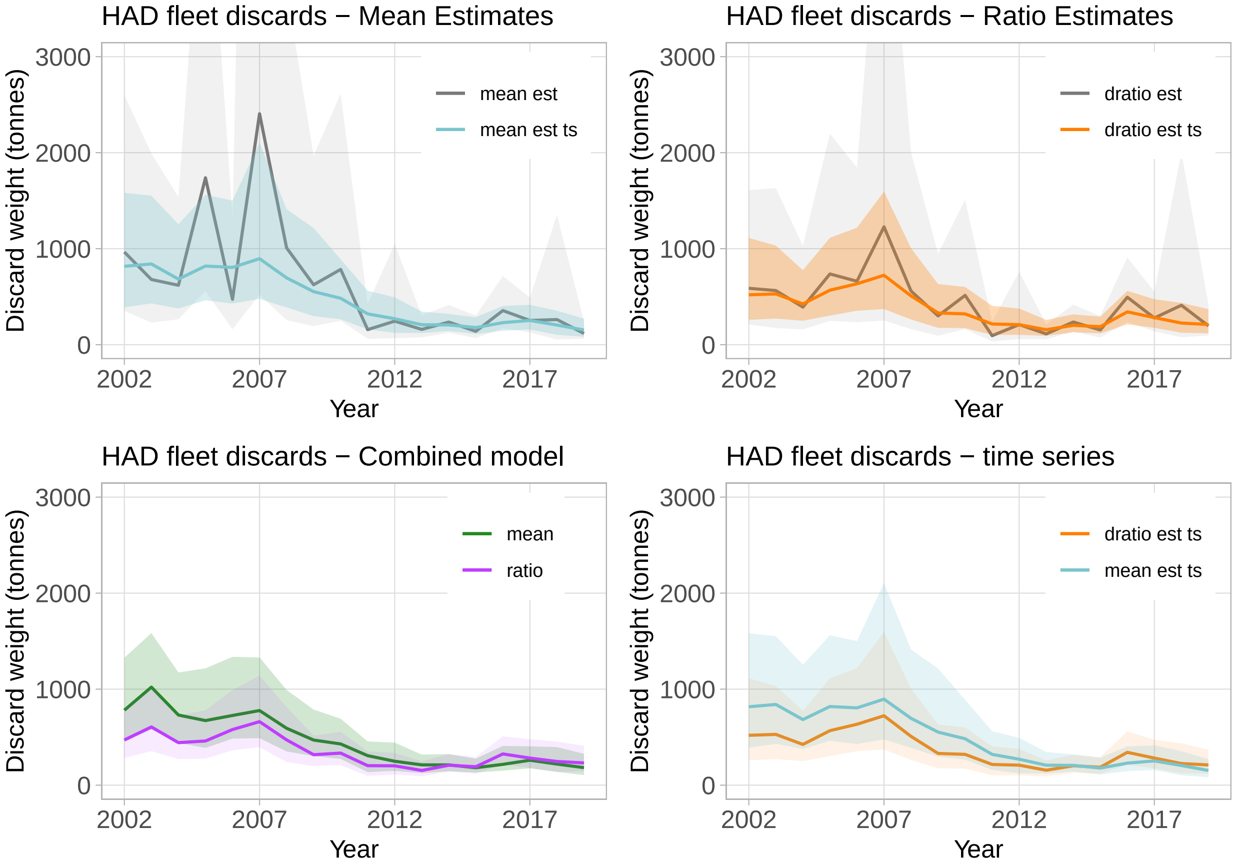

Figure 3.5: Discard estimates calculated at the fleet level using model-based estimators for haddock (HAD). ts represents time series component. mean est - model 1.0, grey line; mean est ts - model 1.1, blue line; dratio est - model 2.0, grey line; dratio est ts - model 2.1, orange line; mean (combined model) - model 3.0, green line; ratio (combined model) - model 3.1, purple line. Shaded areas represent 0.95 credible intervals.

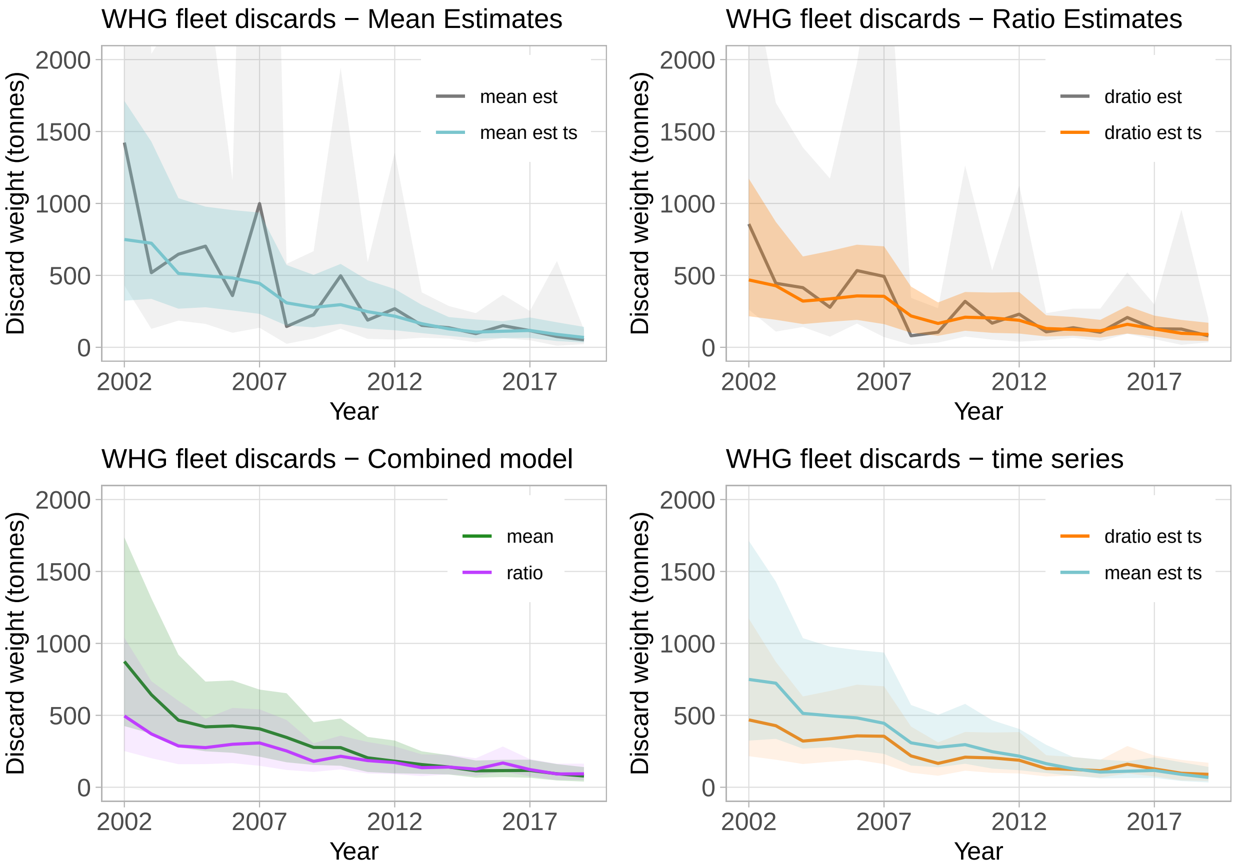

Figure 3.6: Discard estimates calculated at the fleet level using model-based estimators for whiting (WHG). ts represents time series component. mean est - model 1.0, grey line; mean est ts - model 1.1, blue line; dratio est - model 2.0, grey line; dratio est ts - model 2.1, orange line; mean (combined model) - model 3.0, green line; ratio (combined model) - model 3.1, purple line. Shaded areas represent 0.95 credible intervals.

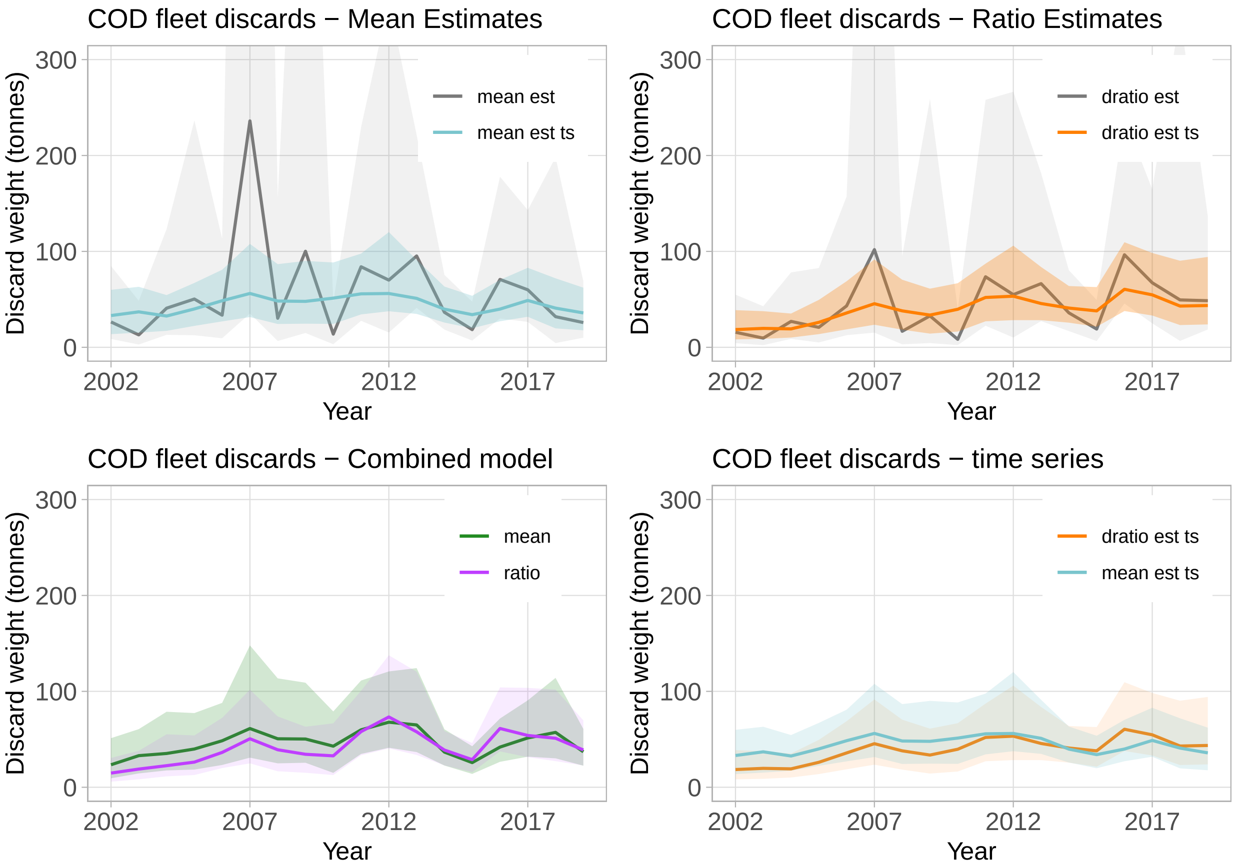

Figure 3.7: Discard estimates calculated at the fleet level using model-based estimators for cod. ts represents time series component. mean est - model 1.0, grey line; mean est ts - model 1.1, blue line; dratio est - model 2.0, grey line; dratio est ts - model 2.1, orange line; mean (combined model) - model 3.0, green line; ratio (combined model) - model 3.1, purple line. Shaded areas represent 0.95 credible intervals.

3.4 Discussion

3.4.1 Comparison of methods to estimate discards

Model-based methods appear to provide more precise estimates of discards when compared with design-based estimators (the latter compute negative values of discards for the 95% confidence intervals), even without including a time series component. An important difference between the design-based methods and the models for the basic mean and ratio estimators is that the models assume a constant variance for all years, reflected by a single parameter representing the observation error on discards, while the design-based methods estimate yearly values for the discards variance. Assuming a constant variance across years might be a reasonable representation of the discard sampling protocol since no major changes are expected to occur on the sampling methodologies from one year to the next provided the sample size is similar. Previous studies on discards in the Clyde used demersal fish landings as an auxiliary variable to raise discards from the observer trips at the fleet level (Stratoudakis et al., 1999; Stratoudakis et al., 2001). However, these studies were conducted between 1982 and 1998 when a reduction of 87% was observed in fish landings, so another variable had to be used for the current regime where no fish is landed. The Nephrops landings which represents the target species of the fishery seems to be an appropriate auxiliary variable, directly correlated with discards weight, as recommended by Fernandes et al. (2011).

Both the discard ratio and the mean estimator reflect the conditions in a specific year of factors such as fish abundance in the sea and fishery regulations (e.g. quota restrictions, bycatch limits). Although the design-based estimators account for variation between trips at the level of the sampling design, the ratio estimator includes another source of variability from measurements of Nephrops landings for every trip. The mean estimates are slightly higher than the ratio estimates for haddock and whiting. However, the estimated discards for cod are similar using the mean or the ratio estimators. The number of fleet trips and the Nephrops landings weights are both measures of fishing effort.

Adding a time series (ts) component to the mean/ratio models appears to greatly improve the estimates of discards (lower DIC and pD). The models containing a ts component have a lower number of effective parameters due to the correlation across years. In fact, it is expected that the values are correlated, because no major changes to fleet behaviour are expected from one year to the next. The ts models have a lower fit to the data, but capture the main trends of discards over time and likely account for observation error.

The combined model relates discard quantities with landings as a measure of fishing effort and a relative index of abundance of fish in the sea given by the survey index. Even though this includes more information and could potentially provide better estimates of discards since the amount of discards should be related with the amount of fish in the sea, this model does not appear to give better results compared with the simpler approaches of the mean/ratio estimators with time series smoothing. One explanation could be the high variability of discard quantities between observer trips, which are both due to measurement error (from the subsampling methodology) and process error. The process error is the result of the natural process of fishing on a patchy population because fish are not evenly distributed across an area. The combined model tries to relate individual trip data from one year to the survey index which represents the average abundance in the Clyde for that same year. However, it seems that the trip data are dominated by the variation within each year which the survey index cannot account for, and thus not being able to relate the biomass signal given by the survey with the discard patterns from year-to-year.

Traditional methods to estimate discards may benefit of a modelling approach, both to reduce uncertainty in the estimates as well as including other sources of data/information. In this work, modelling the discard weight using Bayesian estimation methods allows the estimation of different error sources (measurement error, process error) in the data, and contributes to a better understanding of how variability in discard estimates occurs. It is desirable to communicate the precision/accuracy of the discard estimates. For stock assessment purposes, a coefficient of variation (CV) of no more than 20% is preferred. CVs reported by Stratoudakis et al. (1999) were between 50-60% for the West of Scotland, while Fernandes et al. (2011) estimated lower CVs (30-40%) using bootstrapping methods to calculate uncertainty in the discard estimates. Both studies suggest that to improve precision of estimates, additional sampling should be conducted, which requires more human and financial resources to increase observer effort. The results of this work provide an alternative methodology that improves the precision of these estimates without the need of any additional sampling. The design-based estimators have the advantage of being unbiased, but also assume that one does not have knowledge on the discard rates from one year to the next. The model-based estimators make use of this knowledge with a time series model and greatly reduce the uncertainty in the estimates. However, the discard weights from this methodology are potentially biased. Thus there is a trade-off between precision and bias that merits further investigation to determine which approach leads to the lowest mean squared error (MSE). In this study, given the very low precision of the design based estimators due to the low level of annual sampling, exploiting prior knowledge was considered to be a preferable approach while recognising there may be unwanted bias. Where modelled discard weights are autocorrelated care must be taken with using them as input data for stock assessment models, particularly in the observation error assumptions.

3.4.2 Fleet discards patterns over time

For haddock and whiting, there is a decreasing trend of discarded weight between 2002 and 2019. Two technical measures were introduced on the Nephrops fishery operating in the Clyde between 2008 and 2009 to reduce bycatch quantities and improve the selectivity patterns (Mills et al., 2017). The first was a square mesh panel on the top of the trawl net to allow larger fish to escape by swimming upwards, and the second was an increase on the mesh size from 70 to 80 mm so juvenile and undersized fish could easily pass through a larger mesh without being retained. These regulations could justify to some extent the discard patterns observed for haddock and whiting. For cod, there seems to be an overall flat trend of discards, possibly explained by small amounts of cod being caught both in the crustacean fleet (one order of magnitude less when compared with haddock and whiting) and by the scientific surveys. Another argument could be changes in the sampling scheme, either by changing the way the vessels are being selected for taking observers onboard, or by altering substantially the number of sampled trips per year. If by chance most of the vessels and/or trips selected have a different discarding behaviour, this will be reflected on the final estimates of discards. Fernandes et al. (2011) also reported low levels of discarded cod by the Nephrops fleet in the West of Scotland when compared with discarded proportions of haddock and whiting, except for 2007 and 2009 when a sudden high level of discarded cod was registered as a consequence of changes in regulatory legislation, though this was not observed with the Nephrops fishery operating in the Clyde.

3.5 Conclusions

Fleet discards were estimated using different methodologies, both design-based and model-based methods. Besides providing updated estimates of whitefish discards in the Clyde, this work contributed with a novel model-based methodology that is distinct from previous studies (Stratoudakis et al., 1999; Stratoudakis et al., 2001). This new method contains a time series component, allows integrating alternative sources of data such as the survey information, and provides more precise estimates of discards. In contrast, previously used methods to estimate discards in the Clyde were mainly based on design-based techniques. Choosing the best method to raise discards at the fleet level relies on a trade-off between opting for the most parsimonious model (simplest model with lowest number of parameters) and the best fit to the data (usually corresponding to the most complex model configuration). The most appropriate model in terms of DIC and number of parameters for each species would be either the ratio or mean model with a time series component. For haddock, the ratio+ts appears to be the best model, while for whiting and cod the mean+ts seems to be an equally suitable model. The ratio estimator needs the Nephrops landed weights (accessible for both observer and fleet trips), which are not available for the observer trips before 2002. As a way to standardize the methods used across species, we choose the mean estimator with a time series component as the most appropriate model for the three species, since it requires the least amount of information that needs to be available for raising the sampled discards to the fleet level.

The next chapters include these mean discard estimates in two distinct population modelling approaches to estimate fishing mortality and population biomass levels for haddock, whiting and cod in the Clyde. Further information on how the discard weights were converted to discard numbers at age is available in Appendix B (Supplementary Information).

CHAPTER 4 - Age-structured population model (ASM)

4.1 Introduction

Age-structured stock assessment models are widely used to evaluate stock status by providing measures of stock abundance and fishing mortality and to manage fish stocks (ICES, 2012). The unit stock can be broken down into cohorts (fish born in a given year), and a cohort will contain fish of the same age and more or less the same body size, that will mature at the same time and will likely be found in the same area (Lassen and Medley, 2001). Following the age structure of a population provides a simple process to model the decay of fish cohorts through time. The cohorts are subject to a total mortality, that can be decomposed into a fishing mortality component and a natural mortality component. The fish only become vulnerable to being caught by the fishery once they are recruited to the main area where fishing takes place and are large enough to be retained by the gear. Natural mortality contains all other sources of mortality besides fishing, and this can be predation, disease and density-dependent effects (e.g. cannibalism, competition for resources). Emigration also represents a loss from the population. Typically, larvae and young fish experience extremely high natural mortality rates compared with adult fish (Lorenzen, 2022). Natural and fishing mortality rates lead to exponential decrease in the numbers of fish in a cohort through time (Beverton and Holt, 1957; Lassen and Medley, 2001; Hilborn and Walters, 1992).

Inspection of the survey and catch data compiled from Chapters 2 and 3 gave an indication that the data have a set of particularities that requires a flexible approach for modelling the demersal fish populations in the Firth of Clyde. The Clyde represents a small area when compared with the wider west coast of Scotland, so the sample sizes are relatively small, which increases the uncertainty in the data. There also appears to be very few old fish in the data sets, which could be either because the survey or fishery cannot catch them (due to gear selectivity or the fish not being present where/when the gear operates), or they simply do not exist in enough quantities to be regularly caught. Multiple surveys occur in the Clyde at different times of the year, so the model needs to allow for the integration of more than one survey as a relative measure of fish abundance. The age compositions of the discards were not available for data before 2002. However, there is survey data available from 1985 onward, so the model had to take into account missing catch observations for this historical period. Taking into account these characteristics, a custom model had to be developed.

The current framework of stock assessment modelling relies on integrated models, which can incorporate multiple sources of data, often framed in a state-space model (e.g. TSA; Fryer, 2002; SAM; Nielsen and Berg, 2014; Stock Synthesis; Methot and Wetzel, 2013). This separates the model into two components: a process model and an observation model. The process model describes how the state of the unobserved fish stock abundance and fishing mortality rates at a given time depend on previous states. The observation model describes how the survey and/or commercial catch data depend on the unobserved states (Perreault et al., 2020). This allows for a clear distinction between the process error, that comes from fluctuations of the true population abundance due to biotic or abiotic factors, and the observation error, that arises from the variability related with the sampling methodology when collecting the data.

Commonly used state-space stock assessment models within the ICES framework until recently were the time-series analysis (TSA; Fryer, 2002) and the state-space assessment model (SAM; Nielsen and Berg, 2014). Both use similar basic equations with slightly different terminology. In many age-structured models, fishing mortality is considered as the product of two factors, an age-specific and a year-specific coefficients (Megrey, 1988). The age-specific coefficient accounts for the different effect of the annual exploitation pattern on the age composition of the stock, and the year-specific effect scales exploitation pattern annually (Megrey, 1988). State-space modelling approaches typically assume fishing mortality (F) as a correlated process, where the value of F for one year depends to some extent on the value of F from the previous year.

In this Chapter, an age-structured stock assessment model (ASM) was set up within a Bayesian framework due to its flexibility, capacity to cope with multiple error structures and to deal with missing and zero observations. The ASM uses some of the features of the previously described models (TSA, SAM) such as time series correlation and a separable fishing mortality assumption. The ASM has similarities to other previously published assessment models using statistical software for Bayesian analysis (Cook, 2013; Cook, 2019a; Cook, 2019b).

The ASM was developed and tested throughout this Chapter. The model equations are described in detail, and the prior assumptions set out. The model was tested using simulated data based on haddock Clyde data, and on real data using the official ICES assessment for the whiting stock in the west of Scotland.

4.2 Assessment model

An age-structured stock assessment model (ASM) is used for the analysis, which has similarities to the currently implemented methods used by ICES such as SAM and TSA. The model is set up to account for zeros in the data, because not all age classes are present for all the years (many older age classes have zero observations). The model does not split discards and landings into different components since the fish landings in the Clyde are negligible after 2002, so the total catches represent the discarded fraction.

4.2.1 Population model equations

The model splits the population into age classes and follows the cohorts through time according to the conventional exponential equation:

$$N_{a+1,y+1} = N_{a,y}e^{-Z_{a,y}} \tag{4.1}$$

where a is an index for age and y an index for year. Z represents total mortality and reduces the number of fish at the start of the year. Total mortality is the sum of fishing mortality F and natural mortality M:

$$Z_{a,y} = F_{a,y} + M_{a,y} \tag{4.2}$$

Fishing mortality is separable into an age effect or selectivity at age s_a, and an year effect f_y (Megrey 1988):

$$F_{a,y} = f_y s_a \tag{4.3}$$

Selectivity is the relative fraction of fish at each age captured by the fishing gear, and the year effect is a measure of the overall fishing mortality at fully selected ages (i.e. when s approaches 1). Here we use a gamma curve to characterise selectivity since is offers flexibility in the shape of the selection curve and is commonly used in assessment models (e.g. Cook et al., 2015):

$$s_{a=j-1} = \left(\frac{j}{\beta(\alpha-1)}\right)^{\alpha-1} e^{(\alpha-1-\frac{j}{\beta})} \tag{4.4}$$

where α is the shape parameter, β the inverse scale parameter that determines the width of the distribution, and j is an index for age class, $j = (1, 2, 3...)$. By convention, fish in their first year of life, (i.e. between birth and the end of their first year) are labelled as age 0 even though they may be as old as one year when caught. 0-group fish are indexed by $j = 1$. Adding one to the conventional age, a (since $j = a + 1$), effectively shifts the curve to the left and avoids computing zero selectivity for 0-group fish (e.g. if $a = j = 0$).

Note that the term $\beta(\alpha - 1)$ represents the age (index) at which maximum selection occurs (the "mode") and the curve can be re-parameterised as:

$$s_{a=j-1} = \left(\frac{j}{mode}\right)^{\alpha-1} e^{(\alpha-1-\frac{j(\alpha-1)}{mode})} \tag{4.4.1}$$

We use this parameterisation when fitting the model as the mode is a more natural quantity when choosing priors for the model parameters.

Fishing year effects f_y are modelled as a random walk:

$$f_y = f_{y-1}e^{\epsilon_y} \tag{4.5}$$

and

$$\epsilon_y \sim Normal(0, \sigma_f), y \neq 1 \tag{4.6}$$

where σ_f is the standard deviation of the random walk. This assumes that the annual changes in F are serially correlated and large values of σ_f mean that fishing mortality can have large annual fluctuations.

4.2.2 Observation equations

The catch in numbers of fish (C) is assumed to follow the Baranov equation (Baranov, 1918):

$$C_{a,y} = \frac{F_{a,y} N_{a,y} (1 - e^{-Z_{a,y}})}{Z_{a,y}} \tag{4.7}$$

The survey index (U) is assumed directly proportional to the population in the sea, where the proportionality constant is the product of an age specific selectivity $sel_{a,k}$ and an overall survey catchability q_k (with k representing a subscript for each survey), both constant over time:

$$U_{a,y,k} = q_k sel_{a,k} N_{a,y} e^{-\rho_k Z_{a,y}} \tag{4.8}$$

The term ρ_k is the proportion of the year before the survey occurs and accounts for the mortality occurring during the year up to the sampling time of the survey. These survey indices of abundance are derived from scientific trawl sampling, so a logistic curve is used to describe the selectivity of the gear. This is parameterized in terms of age at 50% selection ($a50$) and an age range (ar) from Graham et al. (2004) as:

$$ln\left(\frac{sel_{a,k}}{1 - sel_{a,k}}\right) = \left(\frac{ln(9)}{ar_k}\right) a - \frac{a50_k ln(9)}{ar_k} \tag{4.9}$$

and thus

$$sel_{a,k} = \frac{1}{1 + exp\left(ln(9)\frac{a50_k - a}{ar_k}\right)} \tag{4.10}$$

4.2.3 Observation error distributions

The observed survey indices, $U'_{a,y,k}$ are assumed to be lognormally distributed with survey and age-specific standard deviation:

$$U'_{a,y,k} \sim lognormal(log(U_{a,y,k}), \sigma^u_{a,k}) \tag{4.11}$$

For the base ASM version, the survey data is included as 2 surveys, survey 1 (spring quarter 1) and survey 2 (autumn quarter 4) surveys. However, there has been a change in the survey design and sampling locations that likely affected the survey efficiency in catching fish between 2010 and 2011 (see Chapter 2 - Survey data). To account for this, an additional error parameter ($\sigma^{add}_{a,k=2}$) was included that allows $\sigma^{u}_{a,k}$ to take higher values from 2011 onwards.

The observed catch (i.e. discards) is observed with lognormal age-specific error σ^{c}_{a}:

$$C^{'}_{a,y} \sim lognormal(log(C_{a,y}), \sigma^{c}_{a}) \tag{4.12}$$

Equations 4.11 and 4.12 are not defined for zero observations and will cause bias in any parameter estimates if zero values are simply omitted from the likelihood. This issue is discussed in the next section.

4.2.4 Hurdle likelihood

The hurdle model is separated in two processes: separating the zero and non-zero data first, and then after a certain "hurdle" is cleared, modelling the non-zero observations with another process.

The observations, $\hat{y}$, comprise values of catch at age ($C^{'}_{a,y}$) and survey indices ($U^{'}_{a,y}$) that are subject to sampling error. For these observations, the assumption is that non-zero values are described by a lognormal density function, $g(\hat{y}|\theta)$, with parameters $\theta = (log(y), \sigma)$ that represent the mean and standard deviation. Here, y is the true value of the quantity of interest (i.e. $C_{a,y}$, $U_{a,y}$).

For zero observations (i.e. $\hat{y} = 0$ for $y \geq 0$) we assume the probability that $\hat{y} = 0$ is given by a logistic function where the probability of observing a zero value decreases with increasing y:

$$p(\hat{y} = 0|y) = 1 - \frac{1}{1 + exp\left(ln(9)\frac{z50 - log(y)}{zr}\right)} \tag{4.13}$$

Here $z50$ represents the value of $log(y)$ when $p(\hat{y} = 0|y) = 0.5$ (midpoint of the sigmoid curve) and zr is a measure of the rate at which the probability declines with increasing y (steepness of the curve). This means there will be an upper bound where the probability of an observation being zero is close to 1, and a lower bound where is close to 0.

The likelihood for all observations can then be written in the form:

$$L(\hat{y}|\theta) = \begin{cases} p(\hat{y} = 0|y) \text{ if } \hat{y} = 0 \\ p(\hat{y} \neq 0|y)\ g(\hat{y}|\theta) \text{ if } \hat{y} \neq 0 \end{cases} \tag{4.14}$$

4.2.5 Stock summary statistics

Standard fish stock summary statistics were calculated within the model estimation procedure to obtain posterior median values and 95% credible intervals of spawning stock biomass, yield, and mean fishing mortality.

The spawning stock biomass (SSB) is defined as the combined weight of all individual fish that have reached sexual maturity and capable of spawning. SSB is used as a measure of the reproductive capacity of a fish stock and its overall status and can be calculated annually as:

$$SSB_y = \sum_a N_{a,y} W_{a,y} mat_a \tag{4.15}$$

with $N_{a,y}$ the number of fish at age per year in the population, $W_{a,y}$ the mean weight at age of an individual fish per year, and mat_a the proportion of fish mature at each age class.

The annual yield or total catches in weight taken by the fishery is given by:

$$Yield_y = \sum_a W_{a,y} C_{a,y} \tag{4.16}$$

where $C_{a,y}$ is the fitted numbers at age for the catch data, summed across ages.

The annual mean fishing mortality is calculated over a minimum ($amin$) to a maximum age ($amax$):

$$meanF_y = \frac{1}{amax - amin + 1} \sum_{amin}^{amax} F_{a,y} \tag{4.17}$$

4.3 Methods

4.3.1 Probability of a zero observation

To test the realism of the assumption that the probability of a zero observation is a logistic function of abundance, we calculated the proportion of zeros at each age across all years. This proportion was then plotted against the log mean abundance of each age across years for both the survey and catch data, using haddock as an example. This provides a simple way of relating the proportion of zeros to abundance since older ages will have much lower abundance on average.

4.3.2 Simulated data

The model was tested on simulated data to demonstrate that the parameters were estimable and to compare the hurdle model configuration with a non-hurdle version. A preliminary fit of the model to the Clyde haddock data was done to generate "true" values; these true values were then used to generate simulated data as described below. The full model fit details to the Clyde haddock data will be presented in the next chapter (Chapter 5).

The Clyde haddock data has been previously described in Chapters 2 and 3. The preliminary fit of the model to the Clyde haddock data used: discarded numbers at age by the Nephrops fishery; numbers of fish at age caught by the Q1 and Q4 surveys; mean stock weights at age; maturity at age data; natural mortality at age data. ICES values of natural mortality for northern shelf haddock were used for this analysis (ICES, 2021b). Given the small sample sizes collected for the Clyde data, is unlikely we have enough information to estimate natural mortality for the fish stocks in this area.

The simulated data was created in steps:

1. The following quantities were extracted from the model fit: fitted values of survey indices and catches, standard deviation of observation error and the probabilities of a zero observation in the data;
2. Pseudo-data values of the observed quantities (survey indices and catches) were drawn from lognormal distributions using the fitted values as means and standard deviation

as observation errors;

3. To simulate zero values, random values were set to zero based on the probability of zero from the model. Pseudo-data values were set as missing (NA) in the same pattern as occurred in the real data.

4.3.3 ICES data

The model was tested by applying it to the west of Scotland whiting stock. The model was fitted using similar data as the ICES assessment, which is the internationally agreed assessment (ICES, 2021a). This consisted of two surveys (Scottish groundfish survey for Q1 and a combined Scottish and Irish groundfish survey for Q4), in numbers of fish for each age class per 10 h of survey effort; catch data for all fleets as numbers of fish at age; smoothed mean weights at age for the stock (from a combination of catch and survey data); fixed natural mortality values and a maturity ogive. A summary of the survey and catch data used are in Table 4.1. All data sets were extracted from ICES (2021b).

Table 4.1 - Data range extracted from ICES (2021b) for the 6a whiting stock.

Data source	Years used	Ages used
Catch data	1985-2019	0-7
Survey 1 (quarter 1)	1985-2019	1-7
Survey 2 (quarter 4)	1996-2019	0-7

The model was configured to account for 2 surveys and 8 age groups, and does not include a plus group (cumulative oldest age group in the population). In the ICES assessment, the Scottish groundfish survey (Survey 1) is included as 2 separate surveys: ScoGFS-WIBTS-Q1, between 1985 and 2010, and UK-SCOWCGFS-Q1, between 2011-2019. For the analysis with the ASM, we included this survey as a single time series from 1985 to 2019.

By fitting our model to a similar set of data, the ICES assessment should provide a reference assessment to be used as means of comparison to better understand how realistic are the outputs from the ASM. If the estimated trends of spawning biomass, recruitment, mean fishing mortality and yield/catches provided by the ASM are in reasonable agreement with the existing age-based stock assessment model, then it provides reassurance the ASM can be applied to the Clyde stocks. The trends in SSB, mean F, recruitment and catches from the ICES assessment for this comparison were extracted from ICES (2021b), where the state-space assessment model (SAM) was used.

4.3.4 Model configuration and model fitting

Priors on the parameters are considered uniform or log uniform. This means that the prior can take any value between the bounds of the uniform distribution (weakly informative prior).

Table 4.2 - Prior distributions on the model parameters.

Parameter	Prior	Description
$log(N_{1,y})$	$Uniform(-10, 20)$	Initial population at youngest age (recruits)
$log(N_{a,1})$	$Uniform(-10, 20)$	Initial population in the first year
f_1	$Uniform(0, 2)$	Initial fishing mortality
σ^f	$Uniform(0, 1)$	Standard deviation of the process error on f
α	$Uniform(1, 20)$	Shape parameter of fishery selectivity
$mode$	$Uniform(2, 4)$	Age (index) of maximum fishery selectivity
σ^c_a	$Uniform(0, 10)$	Measurement error on the catches
$a50_k$	$Uniform(-5, max(j))$	50% selection age from survey selectivity function. If all fish are fully selected at the youngest age, negative values are possible
ar_k	$Uniform(0, 3)$	Survey age range
$log(q_k)$	$Uniform(-10, 10)$	Log survey catchability
$\sigma^u_{a,k}$	$Uniform(0, 10)$	Measurement error for the survey indices
$\sigma^{add}_{a,k}$	$Uniform(0, 10)$	Additional error term to account for change in survey in 2011
$z50_k$	$Uniform(-10, 10)$	Midpoint of the logistic function of probability of zero observation
zr_k	$Uniform(0, 5)$	Steepness of the logistic function of probability of zero observation

The model was fitted using the package ‘RStan’ (Stan Development Team, 2023). As a preliminary fit to the Clyde data, three MCMC chains were run with a minimum of 100 000 iterations, a burn in of 50 000 and a thinning rate of 250. Model convergence was assessed with trace plots and the use of the R-hat value.

For the simulations, thirty simulated data sets were drawn from the true values, and the ASM model was fitted to these in turn. Three MCMC chains were run with a minimum of

100 000 iterations, a burn in of 50 000 and a thinning rate of 250. As well as the full hurdle model, a non-hurdle model was fitted that excluded zero observations from the likelihood to compare performance when these observations are omitted. The mean/median, maximum and minimum of the stock summary statistics were computed and compared with the true values.

The ASM was fitted to the ICES data using the same configuration. Standard summaries of the posterior distributions were calculated (e.g. mean, median, 95% credible intervals). The median values of the parameter posterior distributions were used for plotting and further analysis as these are likely to be more robust to asymmetric distributions.

4.3 Results

4.3.1 Hurdle model results

The expected values of the model were assumed to depend on the probability of a zero observation according to a logistic function (see equation 4.13). There is an approximate indication of this relationship by calculating the mean survey/catches for each age from the data over all years, and plotting it against the proportion of zeros in each age group (Figure 4.1).

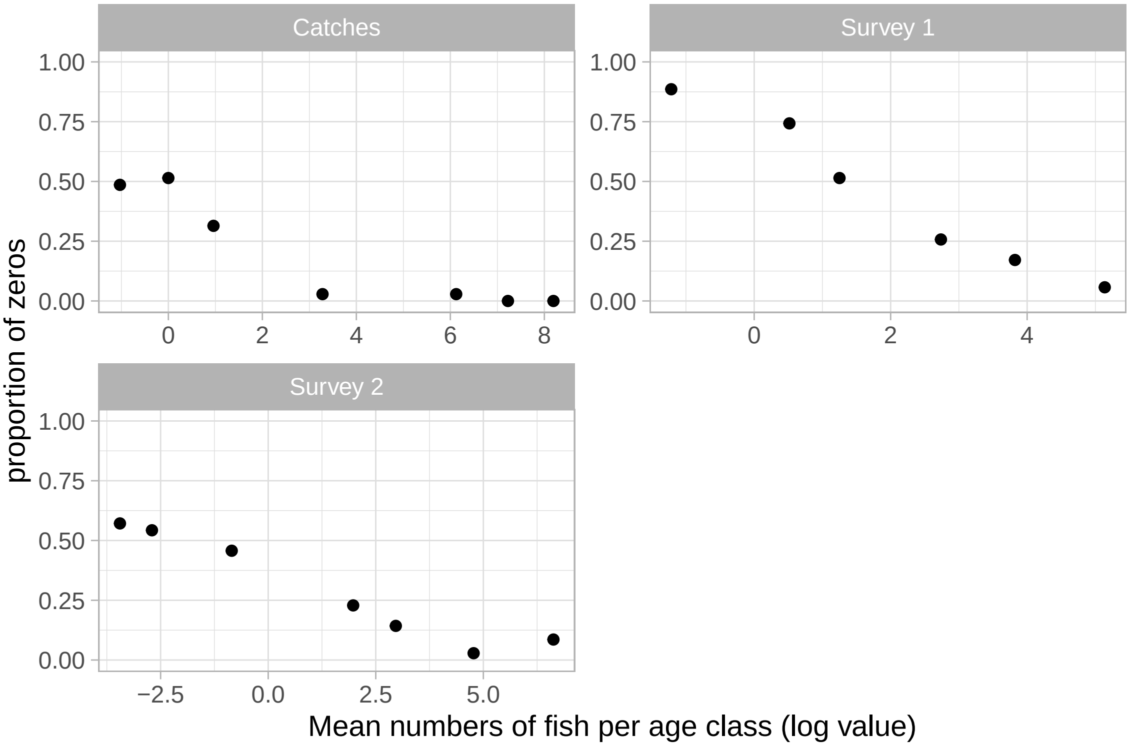

Figure 4.1: Proportion of zeros in the haddock data according to mean values of the catch data (in numbers of fish per age class) and mean survey data (in numbers of fish per hour per age class) calculated across years. Mean values are in logarithmic scale.

When the mean index values approach zero, the proportion of zeros reaches maximum high values. Where the mean index values increase to reach a maximum, the proportion of zeros approaches a minimum bound; this supports a logistic shaped curve. The oldest age group corresponds to the left hand of the plot (highest proportion of zeros), while the youngest age corresponds to the right hand of the plot (lowest proportion of zeros).

The plots shown in Figure 4.2 provide support that the hurdle model configuration has worked as expected. They show the modelled probability of a zero observation against expected values of both survey and catch data, for one simulation. It follows a logistic function, where low expected values of the data will have a higher probability of a zero observation, and vice-versa (higher mean values of the survey or catch data have lower probability of being zero). It seems the model has allocated correctly the zero and non-zero observations to the upper and lower ends of the logistic curve.

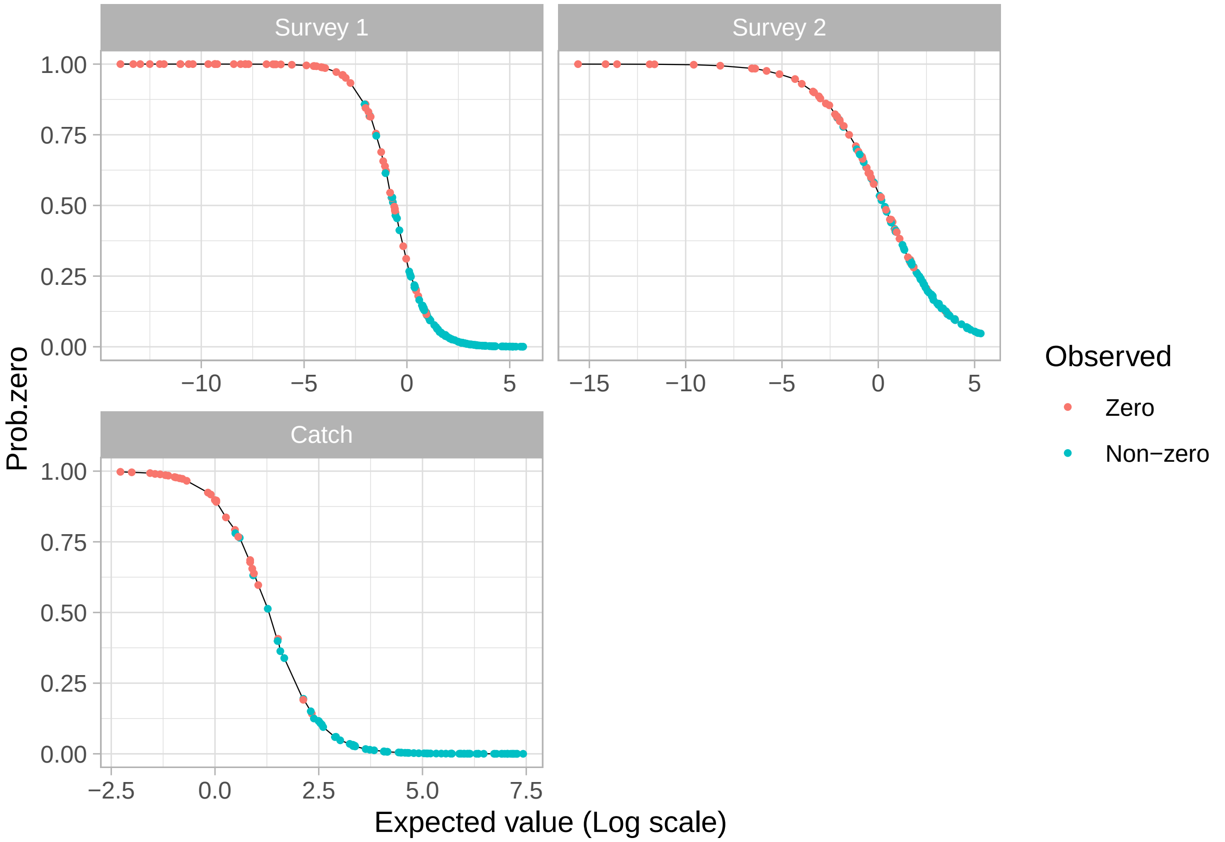

Figure 4.2: Modelled probability of a zero observation estimated for fitted catches and fitted survey values for one simulation. Expected (fitted) values are shown in log scale. NA represents missing data. The coloured points indicate whether the fitted value corresponds to a zero (red) or non-zero (blue) observation.

4.3.2 Simulated data: hurdle vs non-hurdle configurations

A zero observation is not equivalent to a missing value. A missing value refers to an unknown or unobserved value, while a zero value represents an observation that happens to take the value of zero. In the data sets used for this analysis, zero observations occurred when the survey took place or the catch data was sampled, but there were no occurrences of the species of interest. The hurdle model handles zeros with a different process, by explicitly modelling the probability of a zero observation. Figure 4.3 compares the stock summary statistics estimated from both a hurdle and non-hurdle models from the same simulated data set. Both models recovered approximations of the true values from the simulated data. However, the hurdle model recovers values that are very close to the true values, while the non-hurdle model seems to be biased, grossly overestimating the trends in spawning biomass and catches for the early part of the time series (first 5 years of data). The mean fishing mortality is underestimated, not quite at the same level as the true values, and it misses the peak of mean F between 1994-1996. This is because the fishing mortality is considered an auto correlated process within the model, so if the estimated trends in abundance (in numbers of fish) are different at the start of the time series, this has an influence on the pattern of mean F across time. The estimated parameters (α and $mode$) of the fishery selectivity curve are similar between hurdle and non-hurdle models, and both are close to the true values (Figure C.1 in Appendix C). In the non-hurdle model, zeros are treated as missing data and are filled in by the model using the prior assumption. This leads to an overestimation of the biomass and underestimation of the fishing mortality values.

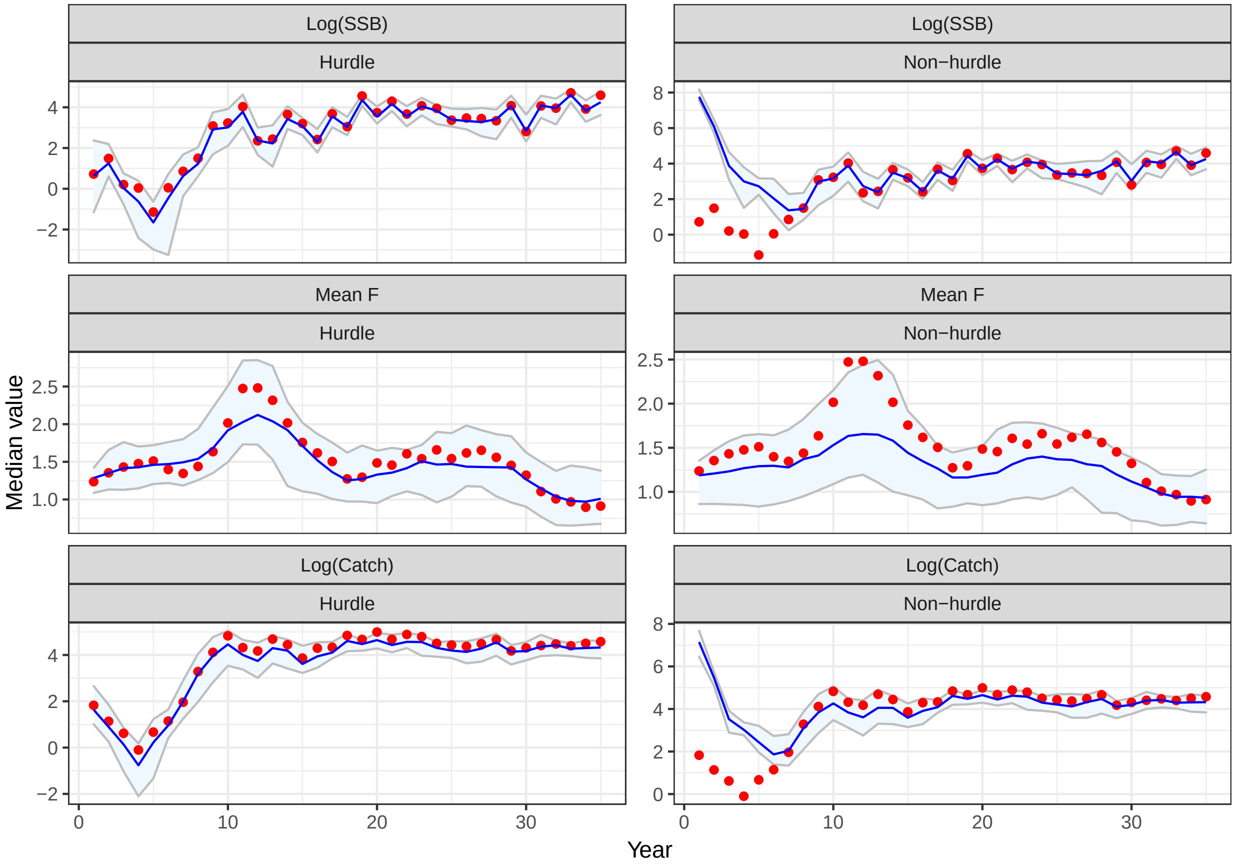

Figure 4.3: Model fit to simulated data, for hurdle model and non-hurdle model. Solid line shows the median of 30 model fits, and the shaded area represents minimum and maximum values. The red dots are the true values.

4.3.3 Comparison with ICES Assessment data

The age-structure model (ASM) produced similar trends in spawning biomass, recruitment, mean fishing mortality and yield as the standard ICES assessment, when using the same data (whiting stock in West of Scotland; Figure 4.4). However, there are some differences between the results of the two models, mostly due to different model assumptions. The ASM tended to estimate lower spawning biomass than the ICES assessment between 1990 and 2015. As a consequence, mean fishing mortality estimated by the ASM seems to be higher, though it starts to overlap after 2010, and from this year onward the ICES assessment estimates slightly higher mean fishing mortality. The recruitment pattern and the total catches are equivalent between the two models, with the only differences being the inter-annual variability of the trends. One difference between these two models is that ASM assumes fishery selectivity at age is constant over time, while SAM allows fishery selectivity to change overtime. Another difference is that the selectivity curve for SAM is dome-shaped, with selectivity decreasing at older ages, while ASM estimates maximum selectivity from age 5 onward. This is shown in Figure C.2 in Appendix C.

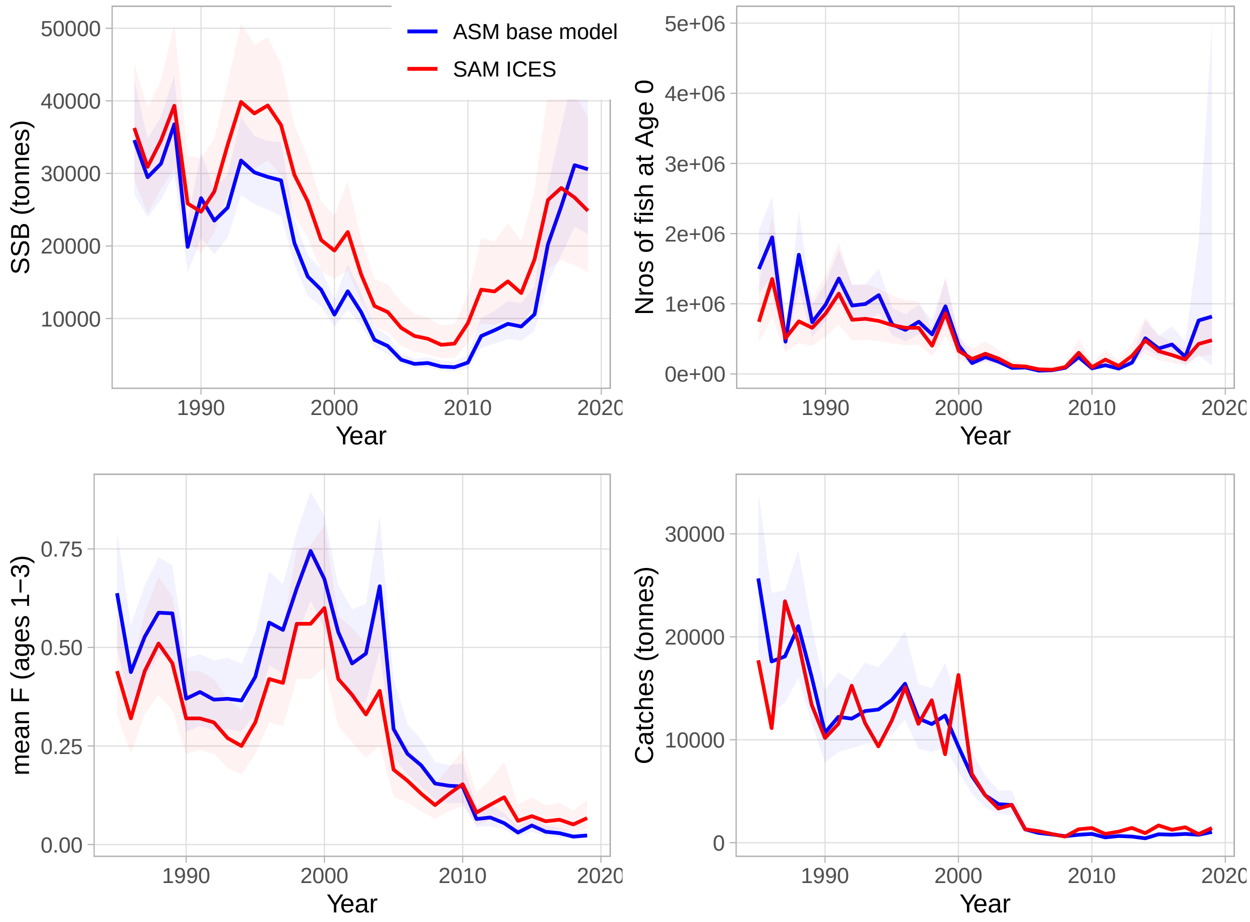

Figure 4.4: Trends for whiting in division 6a (West of Scotland). Spawning stock biomass (SSB), recruitment (nros of fish at age 0), mean fishing mortality (mean F) and catches from the ASM (blue line) and ICES assessment (red line). Shaded areas represent 0.95 credible intervals.

4.4 Discussion

Overall, the age-structured model developed in this work is able to estimate the critical parameters with clearly defined distributions, and these are used to calculate important stock summary statistics such as spawning biomass and fishing mortality.

When testing the model using simulated data, the hurdle version performs better at recovering the true parameter values than the non-hurdle version. The reason for this discrepancy is that the non-hurdle model treats the zero observations as missing data, and fills the zeros with the prior assumption, while the hurdle model treats the zeros with the logistic function of probability of a zero observation. This allows more realism to the model, particularly in the example used for haddock data in the Clyde, where at the start of the time series there are a lot of zeros in the survey data and the catch data is missing. As such, the model does not have a starting point for calculating the absolute abundance of fish in the sea, and the available data indicates that abundance at this period is very low (in contrast to the assumptions of the non-hurdle model). The simulations show that the hurdle model can recover true values, though this is conditioned on these simulated data conforming to the same assumptions as in the original data.

The modelled discard weights from Chapter 3 are used to raise the age compositions and then included as discard numbers at age in the ASM. This creates an inconsistency, since the ASM assumes that the "data" observations are independent, but these are in fact correlated from using a time series model for smoothing the discard weights. This issue likely results in underestimates of variance in the parameters and derived quantities such as SSB. However, the discard weights contribute only to a part of the overall uncertainty in the outputs of the ASM. Other components are uncertainty arising from the age compositions and survey data. In the case of the survey data this issue does not exist, since the observations of the total number of fish caught are independent. Although the estimates of uncertainty are likely to be too low, the median or mean values of the quantities of interest are much less likely to be affected.

When applied to real data used in the standard ICES assessment for whiting in the west of Scotland, the model estimates similar trends in spawning biomass and mean fishing mortality as estimated by ICES. There are considerable differences in scale, and this is likely due to different model assumptions. The ICES data includes three surveys in total, separating the west coast Q1 into two surveys (before and after 2011), and a third survey which combines the west coast and the Irish groundfish surveys. The analysis conducted with the ASM included two surveys only (West Coast groundfish survey of Q1 and combined West Coast and Irish groundfish surveys for Q4, no splitting between years). Including the West Coast groundfish survey as two separate surveys (split between 2010 and 2011) can also influence the trend and scale of estimated SSB. The SAM model includes a plus group, which means fish from a certain age onward (age 7 in this case) are all summed together in the oldest age group. As a result, the oldest age group contains more fish than when including one age class only, and this might have contributed for slightly higher values of SSB in the ICES assessment when compared with the ASM results. Another reason that could explain the differences between the two models is that the SAM models the fishing mortality using another process. It considers a random walk of correlated fishing mortalities, which allows the fishery selectivity to change over time (Nielsen and Berg, 2014). In contrast, the ASM considers the fishery selectivity at age constant for all years (selectivity is age dependent not time dependent), since the Clyde data is too short to realistically estimate changing selectivity over time. This might provide some explanation why the trend of mean fishing mortality is slightly different between the two models. Another aspect is that the ASM does not explicitly include a stock-recruitment relationship. Instead, recruitment is considered random, since the priors on the initial population are uniform priors with wide bounds. This is similar to SAM, where the recruitment process is modelled as a random walk.

4.5 Conclusions

Tests using simulated data show that the hurdle model recovers parameter estimates that have negligible bias compared to the non-hurdle model, at least when the data conform to the modelling assumptions. These tests also show that accounting for zeros is potentially important in reducing bias. When the model was compared to an existing standard assessment model using real data, it produced comparable estimates of SSB and mean fishing mortality. While there were small differences in scale, the trends were very similar. The results suggest that using the ASM is appropriate for the Clyde stocks and this is described in the next chapter.

CHAPTER 5 - Applying ASM to whitefish stocks in the Clyde

5.1 Introduction

This chapter is focused on applying the ASM to three species of demersal whitefish in the Clyde (cod, haddock and whiting). These three gadoid species have similarities between them in terms of life cycles and life history traits. As a reproductive strategy, they are multiple-batch spawners, meaning they produce multiple batches of millions of eggs in one spawning season. The spawning season starts in late winter until spring (Murua and Saborido-Rey, 2003). Cod and haddock migrate to specific spawning grounds where they release their eggs and sperm (ICES, 2005; Casaretto et al., 2014). For whiting, both adult and juvenile stages are found together during spawning season, which might suggest this species does not exhibit spatially distinct spawning areas (Burns et al., 2019). Cod typically reach age at maturity between 2 and 6 years, while haddock and whiting reach maturity at slightly younger ages (2 to 5 years for haddock, and 2 to 4 years for whiting; Froese and Pauly, 2023). As previously reviewed in Chapter 1, there is evidence of dispersal and migration movements at different life stages of haddock and whiting from the Firth of Clyde to adjacent areas. However, for this analysis, the Clyde stocks are treated as single units, and the implications of this assumption are discussed when comparing the results with stock assessments for adjacent areas.

While the previous chapter tested the ASM model, this chapter is focused on applying the model to the three gadoid stocks in the Clyde, while considering species-specific life history traits. Some of the main assumptions of the model were tested by conducting sensitivity analysis, and model comparison was done by visual inspection of results and using the DIC criterion. Analysis of residuals and retrospective patterns provided a visual means of checking the model outputs and test the robustness of the results. Finally, the Clyde results were compared with the assessments conducted for adjacent areas (west of Scotland and Irish Sea).

5.2 Methods

5.2.1 Data

Age-structured abundance indices from survey data and catch numbers-at-age from fisheries data were used for the three species in the Clyde (cod, haddock and whiting). Calculation of these indices was explained in previous Chapters 2 and 3 and a summary of the data ranges used is shown in Table 5.1. Smoothed mean stock weights at age were calculated as described in Chapter 2 for both the survey and discards data. Smoothed mean survey weights from the quarter 1 survey and an average from Q1 and Q4 surveys were used in the analysis for this Chapter. There were almost no fish sampled above age 5, so no plus groups were considered in the analysis.

Table 5.1 - Research vessel surveys and catch data ranges used in the analysis for all three species.

Data source	Years used	Ages used
Catch data	2002-2019	0-5
Survey 1 (quarter 1)	1985-2019	1-5
Survey 2 (quarter 4)	1985-2019	0-4

Maturity data for the three main species in the Clyde was extracted from Hunter et al. (2015). The proportion of mature fish at each age class (Table 5.2) is based on an average between male and female fish, and assumed constant over time.

Table 5.2 - Proportion of fish mature at each age class for haddock, whiting and cod.

Species	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5
Haddock	0	0	1	1	1	1
Whiting	0	0	1	1	1	1
Cod	0	0	0.5	1	1	1

Natural mortality data was extracted for each species from the ICES working group reports:

WGNSSK report 2021 for haddock (haddock in subarea 4, division 6.a and subdivision 20; ICES, 2021b), and WGCSE report 2021 for whiting and cod (whiting in division 6.a; cod in division 6.a; ICES, 2021a).

5.2.2 Base model

A base model similar to the one presented in Chapter 4 was applied, with minor adaptations for each species. The base model had the following assumptions: survey data included as 2 separate surveys (spring Q1 and autumn Q4), with a continuous time series from 1985 to 2019; fishery selectivity at age as a gamma curve; 4 or 5 age groups; natural mortality values extracted from ICES reports (ICES 2021a; ICES 2021b); and smoothed mean weights at age estimated from the Q1 survey only (used for calculating spawning biomass and total catches/yield); 5 age groups were included in the base model for haddock and cod (age 0 to 4), but for whiting only 4 age groups were included (age 0 to 3). Although not representative of the Clyde stocks, the ICES values of natural mortality were used as a starting point for fitting the base model.

While conducting preliminary runs, it was found that the model estimates flat survey selectivity across ages for the whiting data (selectivity = 1 for all age groups). Only the survey catchability parameter (q_k) could be estimated for each survey, so the observation equation for the survey index (equation 4.8 in Chapter 4) was reduced to:

$$U_{a,y,k} = q_k N_{a,y} e^{-\rho_k Z_{a,y}} \tag{5.1}$$

Two different priors than the set of priors previously described in Table 4.2 were used for the runs using the cod Clyde data: $mode \sim uniform(0, max(j))$ and $a50_k \sim uniform(-1, max(j))$, where j is an index for age class ($j = 1, 2, 3...$).

5.2.3 Sensitivity Analysis

For each species, a sensitivity analysis of the base model configuration and assumptions was conducted to understand how changing important model assumptions affected the outputs in terms of biomass estimates and fishing mortality. This provides a way to test the robustness of the model and check if further adjustments are needed, with a view of using the ASM for making predictions about the future.

Six sensitivity scenarios (A-F) were compared against the base model. Each scenario tests one assumption, keeping all the others the same as the base model. The scenarios are described as:

- Model run A: model run with reduced survey data. In this model run, the survey data was included as 2 surveys, but only survey data from 2002 onward was included, in order to achieve complete overlap with the time period of the observer data (2002 - 2019).
- Model run B: model run with fishery selectivity as logistic function. In this model run, we assume a different function for fishery selectivity at age, a logistic curve instead of a gamma curve. The equation 4.4 (from Chapter 4) is replaced by:

$$s_a = \frac{1}{1 + exp(ln(9)\frac{a50_F - a}{ar_F})} \tag{5.2}$$

with $a50_F$ representing age at 50% retention, ar_F the age selection range and a the age class. The following priors were used:

$$a50_F \sim uniform(-5, max(a)) \tag{5.3}$$

$$ar_F \sim uniform(0, 3) \tag{5.3}$$

- Model run C: model run with each survey time series split in 2 surveys. The survey design and sampling locations have changed between 2010 and 2011 (as previously described in Chapter 2), so the survey data can be separated into four time series: before and after 2011, and quarter 1 and quarter 4. This results in four survey indices: Q1 survey before 2011, Q1 survey after 2011, Q4 survey before 2011 and Q4 survey after 2011. The surveys are listed in Table 5.3.

Table 5.3 - Research vessel surveys used in the model run C.

Data source	Years	Ages (haddock and cod)	Ages (whiting)
Survey 1 (Q1)	1985-2010	1-4	1-3
Survey 2 (Q1)	2011-2019	1-4	1-3
Survey 3 (Q4)	1985-2010	0-4	0-3
Survey 4 (Q4)	2011-2019	0-4	0-3

The additional error term ($\sigma^{add}_{a,k=2}$) does not need to be estimated for this scenario (see Chapter 4 - 4.2.3 Observation error distributions), since the survey observation error ($\sigma^{u}_{a,k}$) is estimated separately for each of the four surveys.

- Model run D: model run with one more/one less age group. Some of the older age groups have very few fish or only zeros, so in this model run the influence of including or excluding the oldest age group was tested for each species. The influence of adding one more age group was tested for haddock (age 5 added) and whiting (age 4 added), while for cod the oldest age group (age 4) was removed since there were no fish older than age 4 in the catch data.
- Model run E: model run with natural mortality calculated from mean weight at age, from the results of a meta-analysis of worldwide fish stocks from Lorenzen (1996). It assumes that natural mortality is a function of mean weight at age, $\bar{w}$:

  $$M_{a,y} = M_u(\bar{w}_{a,y})^b \tag{5.4}$$

  where M_u and b are constants that determine the change of M with weight ($M_u = 3.69$ and $b = -0.305$). Smoothed mean weights at age from the Q1 survey were used. Figures D.5, D.10 and D.14 in Appendix D show a comparison of natural mortality values between the base model and model run E.
- Model run F (run only for haddock): model run with alternative estimates of discards. The upper and lower bounds of the 95% credible intervals of the fleet discard estimates

from Chapter 3 were used to investigate how higher/lower values of discard quantities affected the outputs of the model.

Each model configuration was run with three MCMC chains for 100 000 iterations, with a thinning rate of 250. Priors on the parameters were the same as in Chapter 4 (Table 4.2). Posterior means were estimated from 600 random samples from the 3 MCMC chains. Model convergence was assessed with trace plots and calculation of the R-hat diagnostic value. Histograms of the posterior distributions of the critical parameters were checked, and the median values were chosen for further analysis in case the parameters showed skewed distributions.

5.2.4 Model comparison

For comparison of the different model runs, the deviance information criteria (DIC) and the effective number of model parameters (pD) were calculated from a log likelihood function based on 600 random samples from the 3 MCMC chains. This calculation was previously described in Appendix B of Chapter 3 (equations B.8 - B.9). The DIC can only be used to compare models that use the same data. For example, model run A uses less information than the base model, so the DIC cannot be compared between these two model runs. After visualization of model outputs and comparison of DIC between models, a final model configuration was chosen for each species ("reference model"). This reference model takes into account changes in model assumptions tested during the sensitivity analysis that lead to an improved model fit or parameter estimation. Histograms of critical parameters and plots of model fit to the survey and the discards data for the reference model are available in Appendix D. Mean fishing mortality (Fbar) was calculated over ages 1 to 3.

5.2.5 Residuals calculation

The residuals for each observation (r_i) can be calculated as:

$$r_i = log(y_i) - log(\hat{y}_i) \tag{5.5}$$

where y_i represents the observed data and $\hat{y}_i$ are the predicted values. Zero observations were not included in calculation of residuals. We used ordinary residuals because the observation noise is assumed univariate and log-normally distributed, and thus we assumed that the residuals are independent (even though some autocorrelation might be expected). The residuals were calculated based on the reference model. Residuals bubble plots are presented such that each bubble is scaled to the residual's size and colored given its sign (positive in red or negative in blue).

5.2.6 Retrospective Analysis

Retrospective analysis was done using the reference model for each species by leaving out the last year of data, until removing up to 5 years of data from the model, one year at a time. For each species, key outputs such as mean fishing mortality (Fbar, across ages 1 to 3), number of fish at age 0 (as a measure of recruitment), spawning stock biomass (SSB) and yield were compared with the model run with all years of data.

5.2.7 Comparison with adjacent stocks

The results from the reference model for each species were compared to the ICES assessment outputs for the west of Scotland (also known as ICES 6a Division) and Irish Sea (ICES 7a Division). Stock summary statistics were compared, such as spawning stock biomass, recruitment, mean fishing mortality across ages 1 to 3, and natural mortality values. The following table summarizes the ICES reports used for extracting values for these comparisons, and the assessment model used for each stock in ICES.

Table 5.4 - Summary of the ICES reports used for extracting stock statistics for the West of Scotland (6a) and the Irish Sea (7a).

Species	Stock summary stats	Natural mortality	ICES assessment model
Haddock 6a	ICES 2013	ICES 2013	TSA
Whiting 6a	ICES 2021b	ICES 2021b	SAM
Cod 6a	ICES 2021b	ICES 2021b	SAM

Species	Stock summary stats	Natural mortality	ICES assessment model
Haddock 7a	ICES 2021b	ICES 2016	ASAP
Whiting 7a	ICES 2021b	ICES 2016	ASAP
Cod 7a	ICES 2018	ICES 2018	ASAP

The ICES assessment for haddock changed in 2013, grouping the stocks from the North Sea and the west of Scotland as a single population (Northern Shelf haddock), so the summary statistics for the west of Scotland were used only until 2013. For cod in the Irish Sea, the estimates were extracted until 2018 since the assessments thereafter used a surplus production model that integrates distinct population dynamics assumptions compared with the ASM, and does not provide estimates of fishing mortality at age. The values of natural mortality for haddock and whiting in 7a were reviewed and agreed during a workshop in 2016 (ICES, 2016) and have remained valid since then, with the assumption of time-invariant natural mortality rates. For comparing the spawning biomass and recruitment between the three areas, the values were standardized to a reference year (2010 was chosen for this analysis). This overcomes the differences in scale between the areas.

5.3 Results

5.3.1 ASM applied to Haddock data

Model comparison - sensitivity runs The base model for haddock was compared with each of the sensitivity scenarios in Figure 5.1 (base model shown as the grey lines). Model run A contained a reduced time series of survey data, with data from 2002 to 2019 so it would overlap with the same time period of existing observer data. The survey data was included as 2 separate surveys, as in the base model. The model run A results are shown as the pink lines. Summary statistics of the model run A show high degree of similarity to the base model, with spawning biomass, recruitment and mean fishing mortality overlapping values. This means that excluding the earlier part of the time series of the survey does not change the model outputs, and has little influence on the most recent trends in the results.

For model run B, the fishery selectivity functional form was changed to a logistic function, and a different set of parameters estimated. In terms of spawning biomass and recruitment, the trends of this model run and the base model remain similar (run B represented as blue line, Figure 5.1). However, the scale of the mean fishing mortality has changed, with much higher mean F values but still maintaining the same trend. Given that older age classes are mostly composed by zero observations and to account for full selectivity of fish older than age 2, fishing mortality needs to be higher as to explain the low number of older fish in the data.

For model run C, the survey data was separated into 4 time series from 1985 to 2019, as previously indicated in Table 5.3. Spawning biomass and recruitment trends are equivalent between model run C and the base model (run C represented as the orange line, Figure 5.1). The scale of mean fishing mortality for this model run is higher when compared with the base model. This might be related with the model not being able to estimate the survey selectivity for the Q4 post-2011 survey (i.e. model estimates flat selectivity across ages), since fishery selectivity and survey selectivity are estimated relative to each other within the model.

Model run D for haddock includes an extra age group (age 5). The spawning biomass and recruitment time series are similar between model run D and the base model (purple line and grey line respectively, Figure 5.1). The mean fishing mortality values are considerably higher, even though the trend over time stays the same. This is due to the fact that when including an older age group, both survey selectivity and fishery selectivity parameters have changed; the age at 50% retention (a50) for both Q1 and Q4 selectivity increased, and the peak of the gamma curve for the fishery selectivity also shifted sightly to an older age group. This will be reflected in higher values of mean F. Age 5 class is composed by very few observations, mostly caught by the Q1 survey and only zero observations for the catch data. This explains why even though model run D includes an extra age group and mean fishing mortality has higher values, the level of SSB and yield stays the same.

In model run E, natural mortality was derived from mean weights at age calculated specifically for the Clyde data. The SSB and recruitment trends for this run look different from the base model, with much higher values after 2003, and particularly high for the most recent 4 years (2016 - 2019; run D shown as green line in Figure 5.1). A comparison of natural mortality values between the base model and run E is shown in Figure D.5 (Appendix D). The mean fishing mortality values decreased substantially when compared with the base model, while the overall trend remains the same. With higher natural mortality values the model will attribute a lower mortality component to fishing mortality, since the model tracks total mortality (Z), and this results in an increased estimate of spawning biomass.

For model run F, the scale of the catch/discards data was changed by using the upper and lower bounds of the discard estimation results. As expected, this resulted in an increased or decreased scale of spawning biomass and recruitment. The results of this run are included in Appendix D (Figure D.6). There was no difference in terms of mean fishing mortality or the selectivity parameters. This is because the scale of total mortality (and thus fishing mortality) is determined primarily by the survey data, which was kept the same. Changing the scale of the catch data will only affect the absolute abundance of the population, but not any other parameter estimated in the model. Besides, the resulting upper and lower estimates of SSB and yield (red and green dashed lines in Figure D.4) coincide approximately with the 95% credible interval of the estimates from the base model. These outputs provide a confirmation that changing the scale of the catch does not affect its trend and does not change the values of fishing mortality. This scenario was run using the haddock data, but it was not considered relevant to repeat it using the whiting or cod data as similar changes of scale are expected without affecting trends in SSB or the values of mean F.

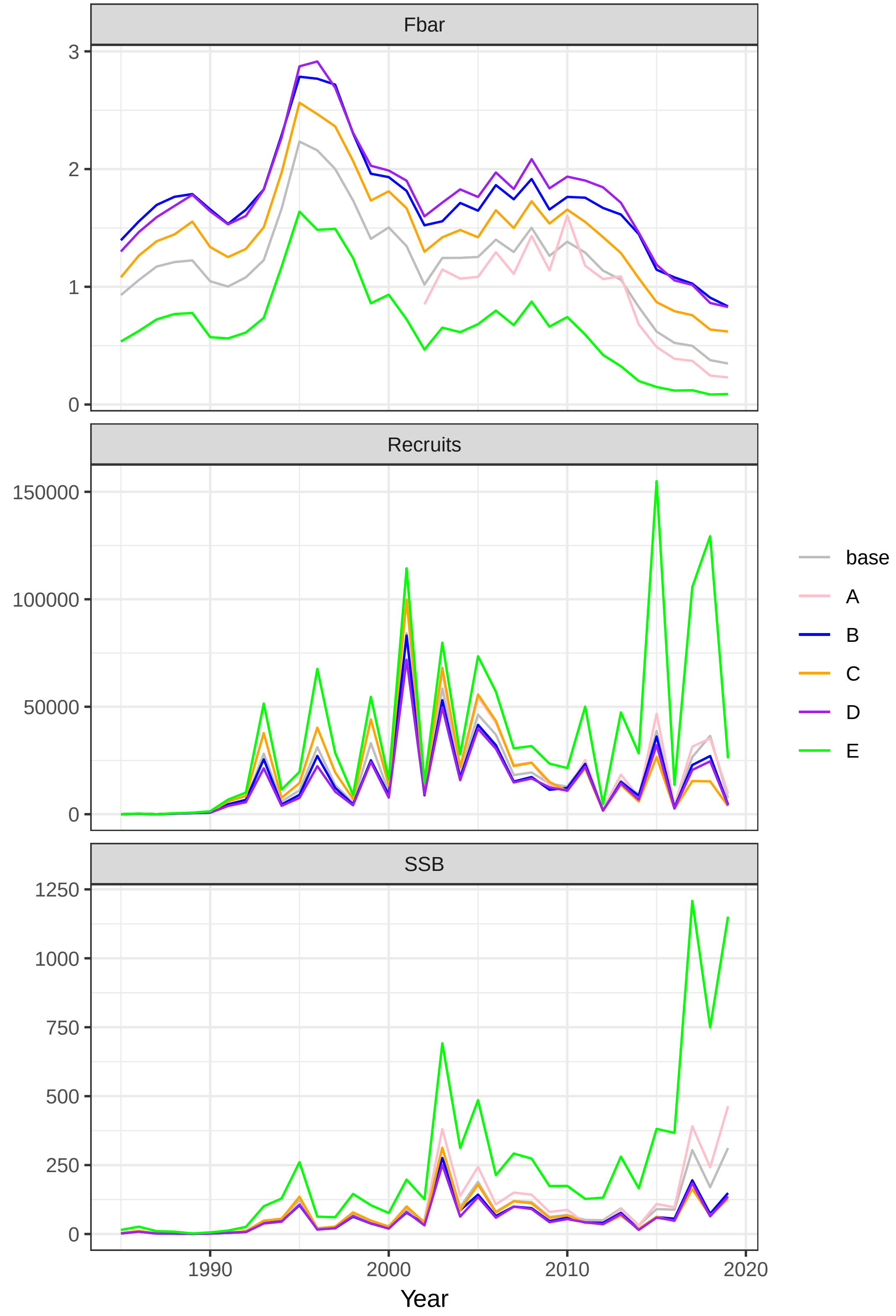

Figure 5.1: Summary statistics for the sensitivity runs applied to Clyde haddock data. Fbar represents mean fishing mortality, recruits are numbers of fish at age 0, SSB is spawning stock biomass in tonnes. The sensitivity scenarios are: base model; A - reduced survey data; B - fishery selectivity as logistic curve; C - 4 surveys; D - one more age group; E - different M assumption.

Model comparison - DIC The DIC and the effective number of parameters (pD) provide another approach for model comparison, although cannot be used to compare all the sensitivity scenarios (Table 5.5). The model run B (fishery selectivity as logistic curve) does not improve the DIC criteria, compared with the DIC value for the base model. On the other hand, model run C has improved model fit to the data and this is reflected in a substantial reduction of the DIC (decrease of 38.41 DIC units). There was a modest increase on the number of parameters estimated for run C. After partition of the survey data into four time series as opposed to two, there were more parameters estimated overall (catchability and selectivity parameters for each survey). Model run E also results in a decrease in DIC compared with the base model.

Table 5.5 - Model comparison (base model, runs B, C and E) of DIC and pD for haddock data. run B - fishery selectivity as logistic curve; run C - 4 surveys; run E - different M assumption.

model	DIC	pD
base model	2639.95	65.45
run B	2652.86	62.69
run C	2601.54	68.38
run E	2619.40	64.89

Comparing all sensitivity scenarios, the following changes to the base model are worth considering for further analysis and for reaching a final model configuration. Changing the assumption on the functional form of fishery selectivity does not provide any improvement in model fit (run B), and the model is not successful in estimating the parameters (model estimates bi-modal distribution of the survey selectivity parameters; Figure D.4 in Appendix D), so the gamma curve for fishery selectivity will be kept. The model run with survey data included as four separate time series (run C) improves substantially the model fit (lower DIC value). Although the model does not properly estimate the selectivity parameters for the post-2011 survey, the selectivity will be fixed for this survey (i.e. no selectivity parameters estimated), and this model configuration will be retained for the reference model. For run D, the age 5 group contains very few observations, so this run will not be retained for the final model configuration. Calculating the natural mortality from smoothed mean weights specific to the Clyde data (run E) is a more realistic assumption than relying on natural mortality values for other areas.

Reference model - Final model configuration The final model, named reference model, had the following configuration: survey data included as four time series, with fixed selectivity as 1 for the Q4 post-2011 survey since the model is not able to estimate it; fishery selectivity as a gamma curve; 5 age groups (0 to 4 year old fish); two separate sets of values for the smoothed mean weights at age, weights from the Q1 survey to estimate SSB (start of the year), and average weights from Q1 and Q4 surveys to estimate annual yield. Sensitivity run E with M based on weight at age gave improved values of DIC compared to the base model that used conventional ICES values. However, run E used Q1 weights that are less likely to characterise M throughout the year. It was therefore decided to use mean weights from Q1 and Q4 surveys to calculate M from the Lorenzen equation. These values are intermediate between the those used in the base model and run E (Figure D.5 in Appendix D) which will be reflected in the estimates F and SSB accordingly.

The summary statistics and selectivity curves for the haddock reference model are shown in Figure 5.2. The results for the reference model are similar to the base model, but provided a much improved model fit (Table 5.6).

Table 5.6 - Model comparison (base model, reference model) of DIC and pD for haddock data.

model	DIC	pD
base model	2639.95	65.45
reference model	2582.73	62.93

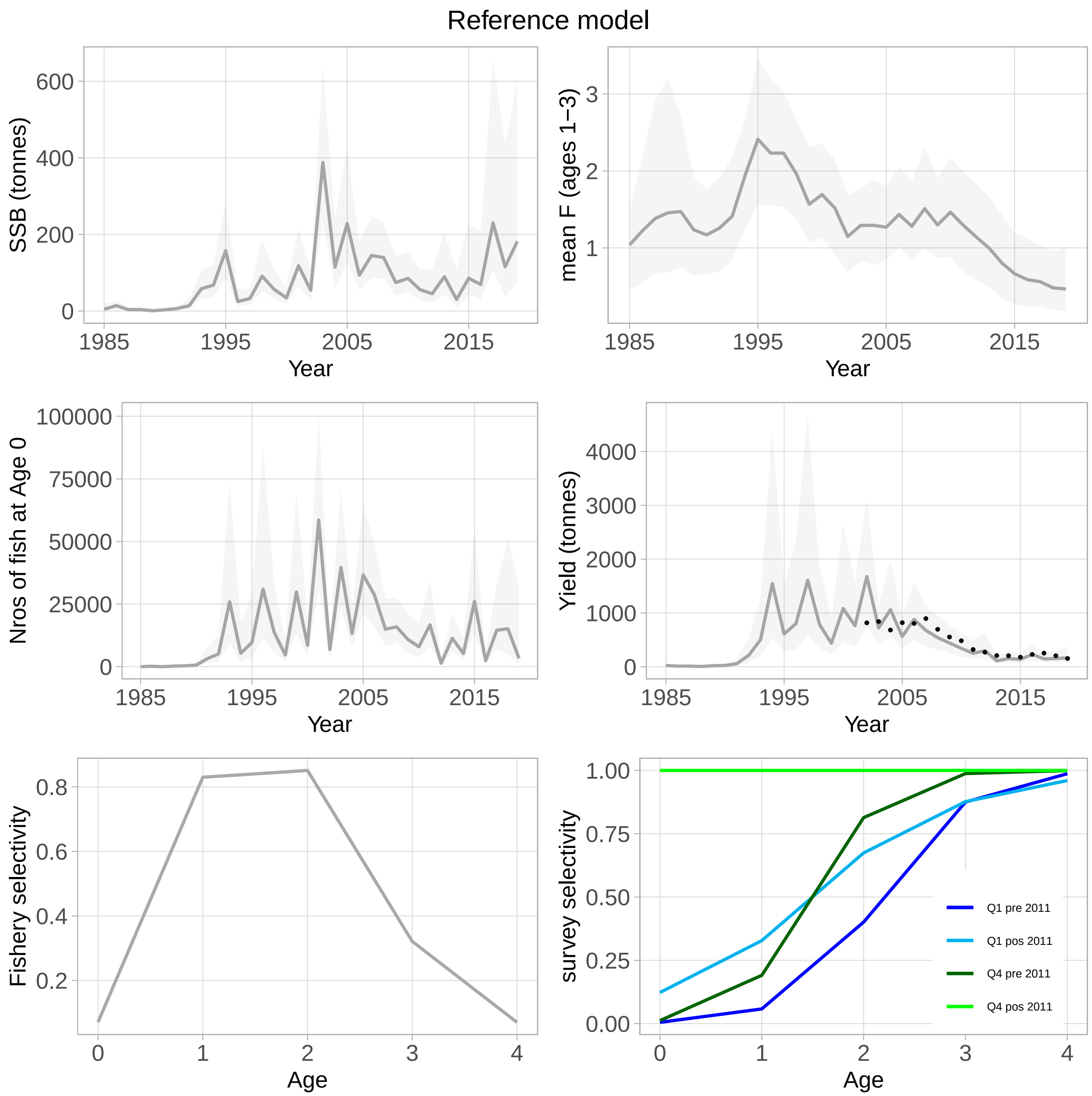

Figure 5.2: Summary statistics and selectivity curves for the reference model applied to Clyde haddock data. Dots represent discards data, shaded areas represent 0.95 credible intervals.

Having reached a reference model configuration that is suitable for the haddock data, the specific patterns of spawning biomass and fishing mortality over time in Figure 5.2 deserve a closer examination. Spawning biomass for haddock reached a peak in early 2000's, decreased thereafter, and in recent years there seems to be an increasing trend. This is likely to be related with the recent peaks in recruitment. In terms of fishing mortality, the model estimates a peak in mean F around 1995. However, there is no catch data included in the model at this period to support this evidence, and it does not seem realistic to expect a two-fold variation in mean F from one year to the next, so the historical trend needs to be interpreted with caution. Fishing mortality has decreased substantially since 2010, and this is associated with the recent increases in spawning biomass and relatively stable low level of yield. The dots in the yield plot represent the total discard weights estimated at the fleet level from Chapter 3. These overlap with the values of estimated yield from the model. The survey selectivity for both the Q1 and Q4 surveys show a sigmoid curve, while the fishery selectivity reaches a maximum selectivity at age 2 that subsequently decreases at older ages.

Residuals checking The residual plot in Figure 5.3 does not show major patterns over years or across ages. There does not seem to be any specific trend in terms of the direction of the residuals, with negative and positive residuals being equally interspersed. The magnitude of the residuals is higher for both Q1 and Q4 surveys post-2011 (survey 2 and survey 4 in the bubble plot). This may be because the most recent surveys seem to have higher observation errors due to the change in the survey design. For the catch residuals, there are larger residuals for the older ages (age 3 and 4) and for some years in age 0. The age 0 class is highly variable, particularly for years with large year classes, due to the increased natural mortality that juvenile fish suffer (Figure D.5 in Appendix D). For the older age classes, this might be because there are very few fish older than age 2 in the data sets.

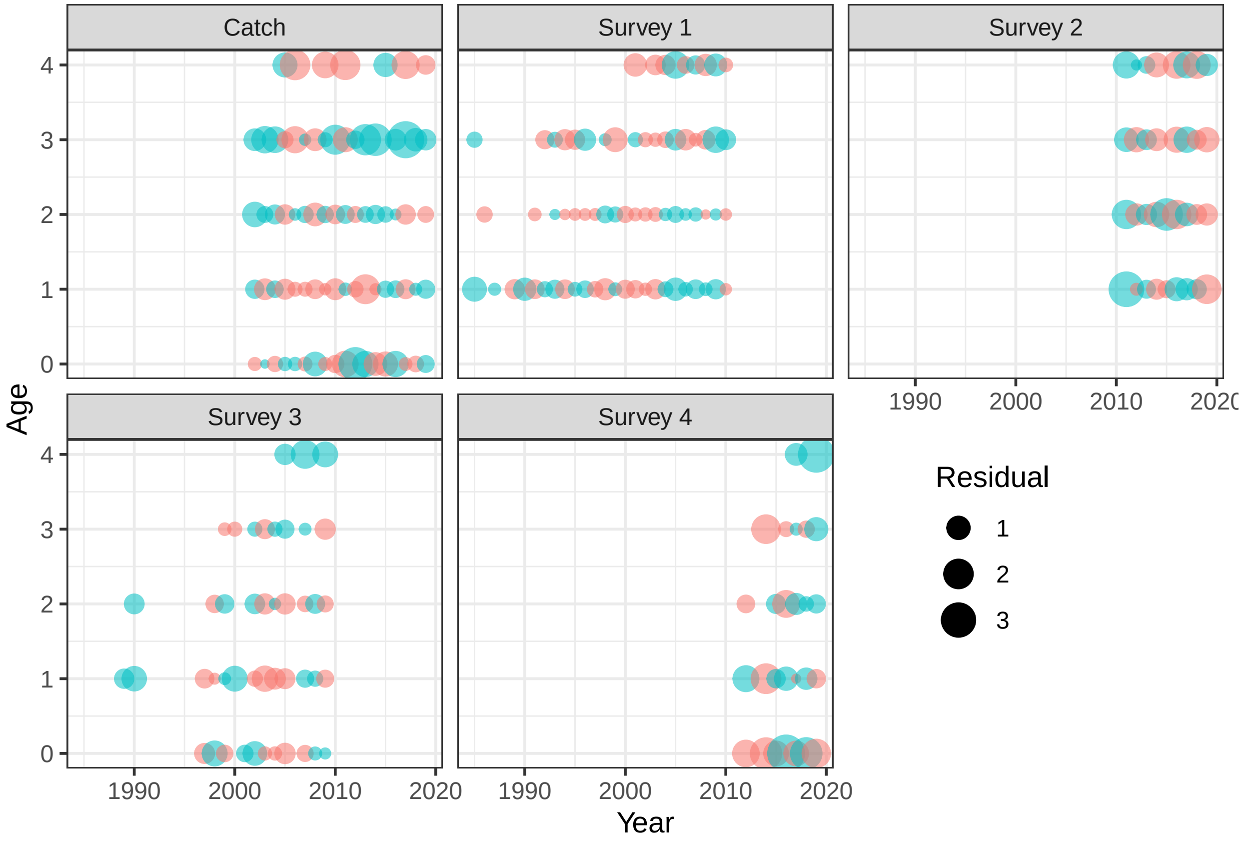

Figure 5.3: Residual bubble plots by fleet from the reference model applied to haddock data in the Clyde. Catch residuals, surveys 1 - 4 residuals. Survey 1 - Q1 survey 1985-2010; Survey 2 - Q1 survey 2011-2019; Survey 3 - Q4 survey 1985-2010; Survey 4 - Q4 survey 2011-2019. Blue = negative residuals, Red = positive residuals. Figure legend shows magnitude of residuals.

Retrospective analysis The retrospective runs for the reference model are shown in Figure 5.4. In terms of mean fishing mortality (Fbar), only one of the model runs with data from 1985 to 2015 (purple line) lies outside the 95% credible interval, and this run also has higher overall values of mean F. For the recruitment and yield time series, excluding previous years does not result in major change of trends. As expected, the most recent upsurge in SSB between 2016 and 2019 is not captured if the last 5 years of data are left out of the model. The estimates of mean F have higher variability between model runs than the estimates of SSB, recruitment or yield. Overall, the exclusion of up to 5 years of the most recent data does not result in major historical revisions to most of the important stock summary statistics (spawning biomass and recruitment), with only one run with a major revision in mean fishing mortality, showing that the model is robust enough to the removal of these data in previous years.

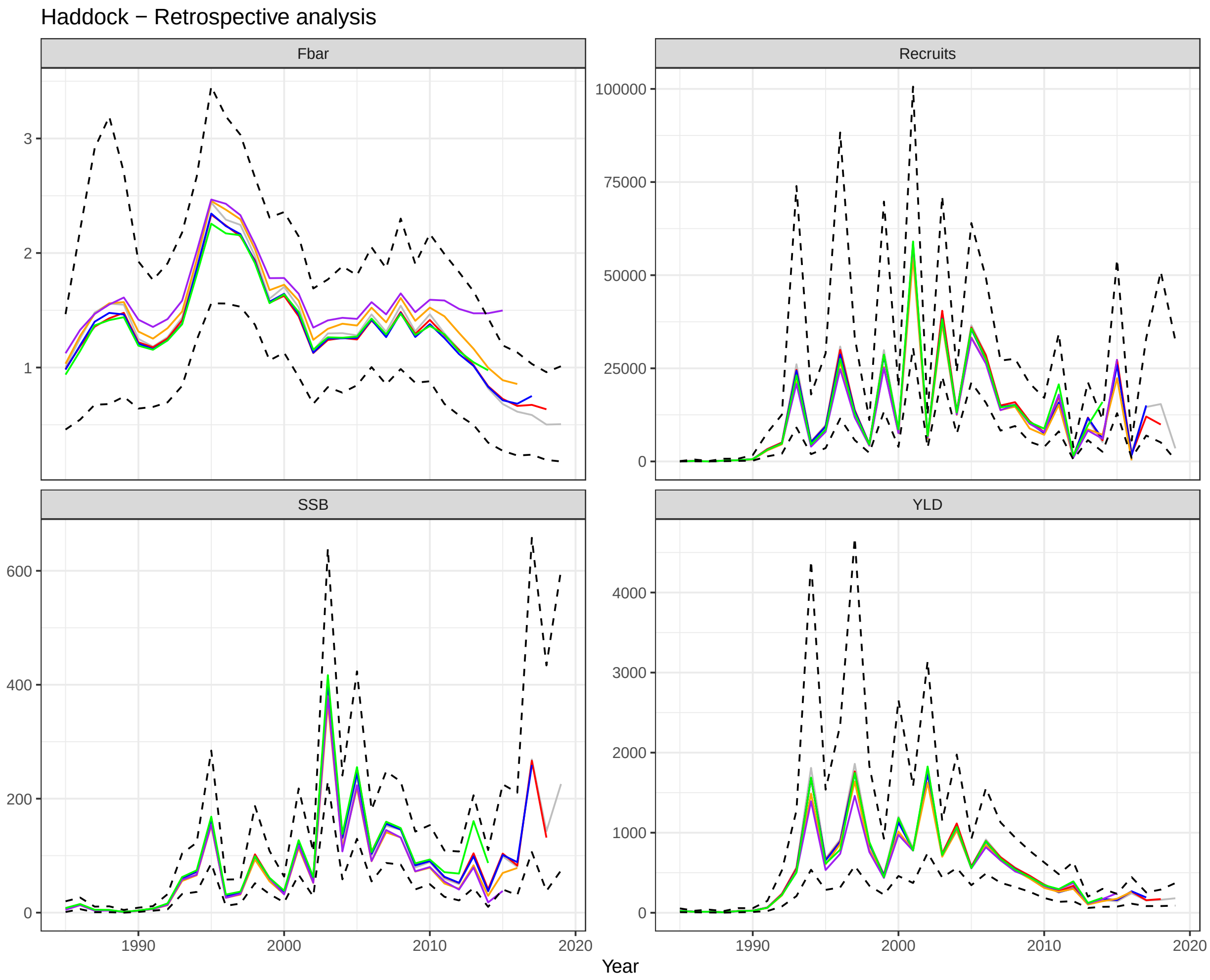

Figure 5.4: Retrospective plots for the reference model applied to haddock data in the Clyde. Dashed lines represent 0.95 credible intervals of the model run containing all years. Fbar is mean fishing mortality, recruits are in numbers of fish at age 0, SSB is spawning stock biomass (tonnes) and YLD is total catch or yield (tonnes).

Comparison with adjacent stocks Summary statistics for the Clyde haddock population were compared with adjacent haddock stocks (West of Scotland and Irish Sea) in Figure 5.5. The latter were extracted from ICES assessment reports (ICES, 2013; ICES, 2016; ICES, 2021b). The mean fishing mortality values estimated for the Clyde are considerably higher when compared with the values for the Irish Sea and west of Scotland. The most recent values for the Clyde are estimated at around 0.5, while for the Irish Sea mean F is below 0.2.

Nevertheless, the trend in mean F is similar across the three stocks, particularly after 2000, where mean F steadily decreases to minimum historical values until 2019. The spawning stock biomass (SSB) and recruitment are represented as standardized values in relation to a reference year (2010 was chosen in this case). The model estimates almost non-existent biomass in the Clyde at the start of the time series, with haddock increasingly appearing from the early 1990's. The increase of biomass and catches after 1990 might be coincidental with the time that haddock seems to have surged in the Firth of Clyde, originated probably from adjacent areas, and the survey data that was used in the model still captures that trend (the model does not include catches before 2002).

There has been a recent increase of SSB in the Clyde, which matches the increased trend of biomass in the Irish Sea, and the beginning of the increasing trend in the time series for the west of Scotland. This recent increase in biomass is probably related with recent peaks in recruitment for both the Clyde and the Irish Sea. The natural mortality values used for the Clyde show a slightly increasing trend. This is because natural mortality is being estimated from the mean weights at age, which are decreasing over the historical time period (Figure 2.7 shown in Chapter 2). The natural mortality values used for the Clyde coincide with the values used for the Irish assessment in the recent period.

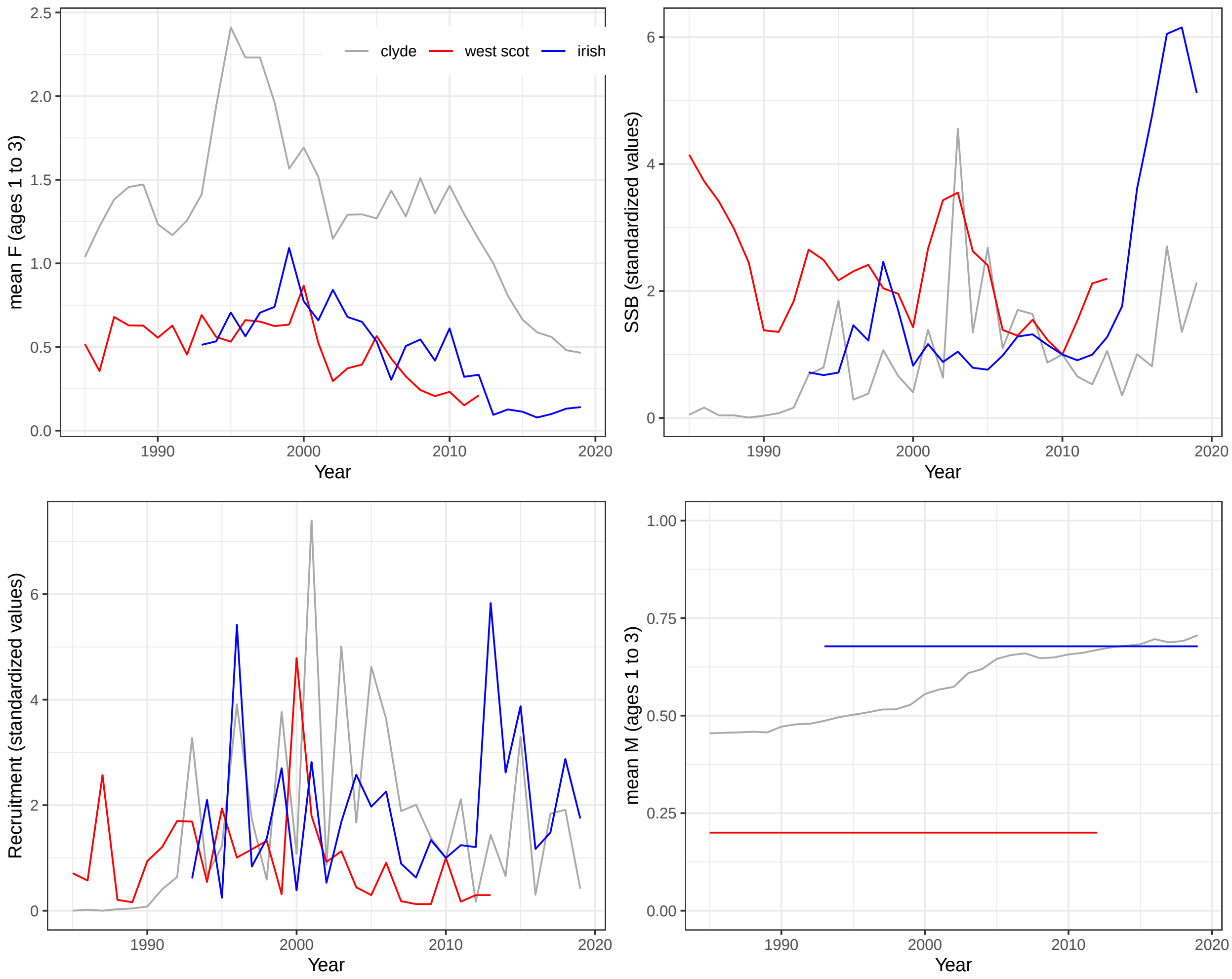

Figure 5.5: Comparison of summary statistics estimated for the Clyde haddock stock with the Irish Sea and west of Scotland areas. Mean F represents mean fishing mortality, SSB is spawning stock biomass, mean M is mean natural mortality. SSB and recruitment values were standardized to 2010 values.

5.3.2 ASM applied to Whiting data

Model comparison - Sensitivity runs The results of the base model are shown in Figure 5.6, and compared with all sensitivity runs (grey line in the plots). Model run A contained a reduced time series of survey data, with data from 2002 to 2019 so it would overlap with the same time period of existent catch data. Summary statistics of model run A show a high degree of similarity to the base model, particularly in terms of spawning biomass and recruitment (run A shown as the pink lines). The mean fishing mortality is slightly lower when compared with the base model. Including only the most recent data does not substantially change the model results, providing confirmation that the most recent trends of SSB and mean F are not sensitive to the early data.

For model run B, the fishery selectivity functional form was changed to a logistic function, and a different set of parameters estimated. In terms of spawning biomass and recruitment, model run B and the base model follow similar trends (run B shown as the blue lines, Figure 5.6). The level of the mean fishing mortality has changed to slightly higher values, but still closely resembling the base model.

For model run C, the survey data was separated into 4 time series from 1985 to 2019. Spawning biomass and recruitment trends are similar to the base model, but the scale has changed to higher values (run C shown as the orange lines, Figure 5.6). The peaks in SSB between 1999 and 2001 are higher for this model run. This might be explained by an improved model fit to the survey data, allowing the model to capture slightly different trends in the data. Mean fishing mortality pattern shows some differences in trend and scale compared to the base model.

Model run D for whiting includes an extra age group (age 4). The spawning biomass trend is similar to the base model, but the peak in recruitment around 1999-2000 is higher for run D (run D shown as the purple lines, Figure 5.6). This might be due to the inclusion of an extra age group, for which there are observations mainly for the Q1 survey, and a few data points for the catch data that might change the model fit to the data. Mean fishing mortality shows a slightly different pattern across time, but still overlaps with the values for the base model.

In model run E, natural mortality was derived from mean weights at age calculated specifically for Clyde whiting data. The SSB and recruitment trends look similar to the base model, but with higher values (run E shown as the green lines, Figure 5.6). The mean fishing mortality has slightly lower values, probably due to the higher natural mortality values used in run E compared to the base model. Higher natural mortality attribute a lower mortality component to the fishery, since the model tracks down total mortality (Z) of the fish cohorts. Higher natural mortality will result in a lower level of fishing mortality and consequently an increased population abundance.

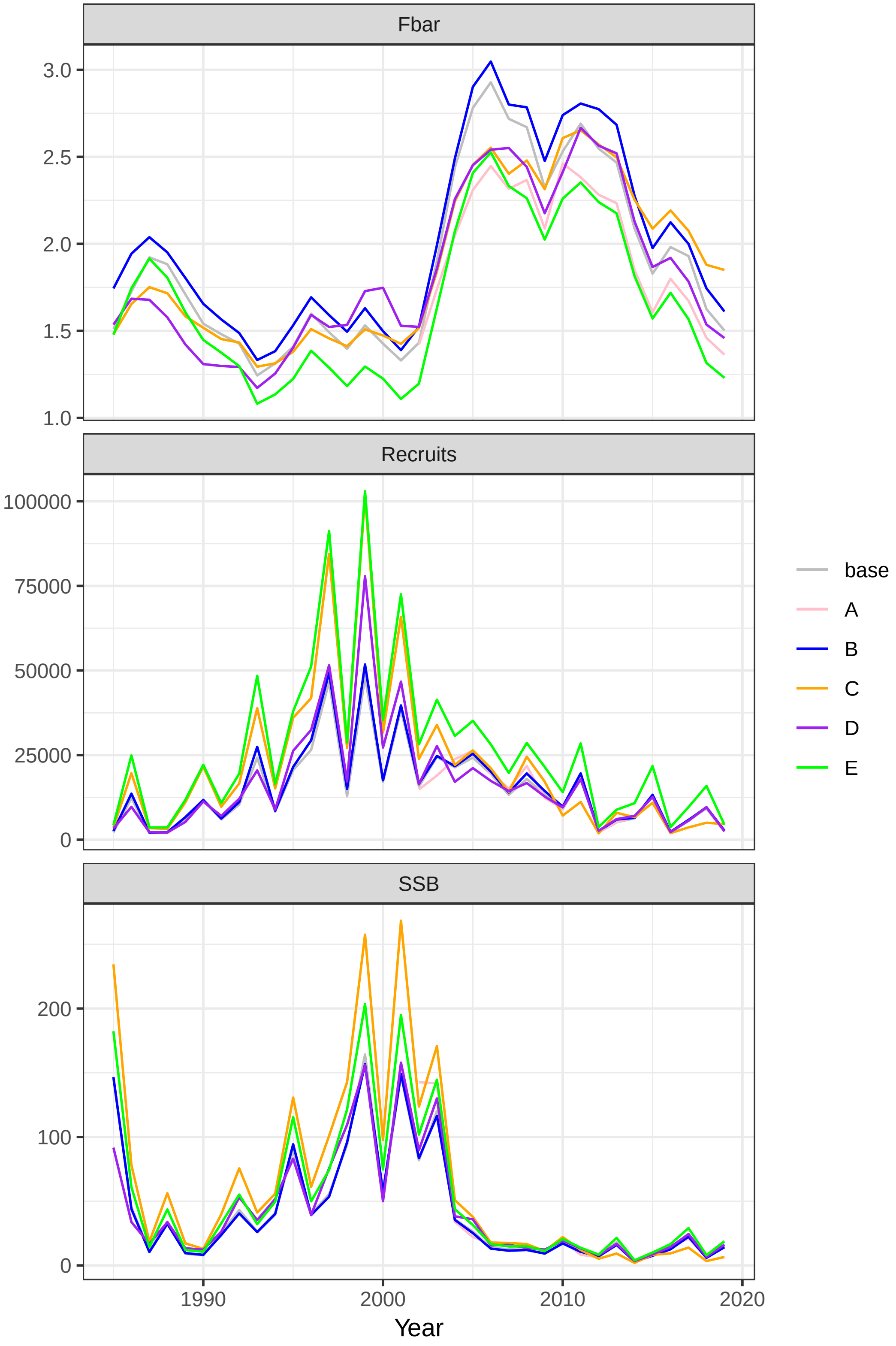

Figure 5.6: Summary statistics for the sensitivity runs applied to whiting Clyde data. Fbar represents mean fishing mortality, recruits are numbers of fish at age 0, SSB is spawning stock biomass in tonnes. The sensitivity scenarios are: base model; A - reduced survey data; B - fishery selectivity as logistic curve; C - 4 surveys; D - one more age group; E - different M assumption.

Model comparison - DIC The DIC and the effective number of parameters (pD) provide another approach for model comparison, although cannot be used to compare all the sensitivity scenarios (Table 5.7). Model run B (fishery selectivity as logistic curve) improves slightly the DIC, but only a difference of 2 units compared with the base model. Model run C (4 surveys) has improved the model fit to the data, and this is reflected in a substantial reduction of the DIC of almost 28 units. This model run, despite separating the survey into four time series, keeps approximately the same number of parameters because the model does not estimate the survey selectivity parameters for the whiting data (see equation 5.1). Model run E does not result in a decrease of the DIC criteria.

Table 5.7 - Model comparison (base model, runs B, C and E) of DIC and pD for whiting data. run B - fishery selectivity as logistic curve; run C - 4 surveys; run E - different M assumption.

model	DIC	pD
base model	3186.03	58.41
run B	3183.39	57.92
run C	3158.33	58.37
run E	3186.48	57.36

Comparing all sensitivity scenarios, there are some changes to the base model that are worth considering for deciding on a final model configuration. Changing the assumption on the functional form of fishery selectivity does not seem to substantially improve the model fit (run B), so the gamma curve for fishery selectivity will be kept. The model run with survey data included as four separate time series (run C) improves substantially the model fit (lower DIC), so this configuration will be retained. For the model run with an extra age group (run D), there seems to be enough information for the age 4 group coming from the Q1 survey, despite the amount of zero observations in the data, so the final model should include this age group. This run could not be compared with the others in terms of DIC and pD. Calculating the natural mortality from mean weights at age specific to the Clyde whiting data (run E) is a more realistic assumption than relying on values for other areas.

Reference model - Final model configuration The final model, named the reference model, had the following configuration: survey data included as four time series, with fixed selectivity as 1 for the Q4 post-2011 survey since the model is not able to estimate it; fishery selectivity as a gamma curve; 5 age groups (0 to 4 year old fish); two separate sets of values for the smoothed mean weights at age, weights from the Q1 survey to estimate SSB (start of the year), and average weights from Q1 and Q4 surveys to estimate annual yield. Unlike the results for haddock, sensitivity run E with M based on weight at age gave similar values of DIC compared to the base model that used conventional ICES values. However, for consistency with haddock and cod, mean weights from Q1 and Q4 surveys were used to calculate M from the Lorenzen equation. These values are intermediate between the those used in the base model and run E (Figure D.10 in Appendix D) which will be reflected in the estimates F and SSB accordingly.

The summary statistics and fishery selectivity curve for the reference model are shown in Figure 5.7. The reference model can be compared in terms of DIC and pD to run D, since both include the same data used in a slightly different configuration. The reference model improves the model fit substantially, with a reduction of the DIC of almost 31 units (Table 5.8).

Table 5.8 - Model comparison (run D and reference model) of DIC and pD for whiting data.

model	DIC	pD
run D	3374.06	60.65
reference model	3343.11	59.29

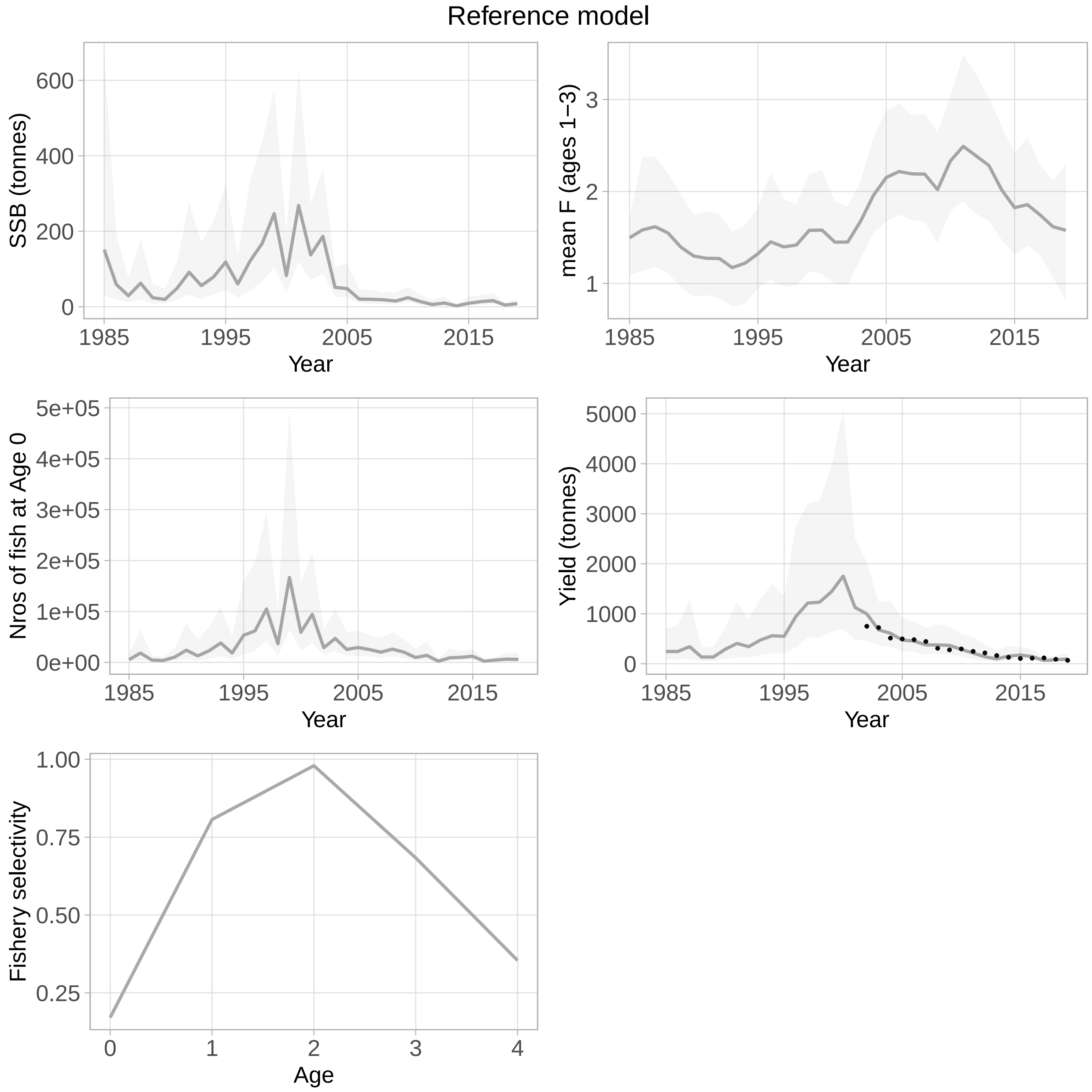

Figure 5.7: Summary statistics and selectivity curve for the reference model applied to Clyde whiting data. Dots represent discards data, shaded areas represent 0.95 credible intervals.

Looking at the specific patterns of biomass and fishing mortality over time in Figure 5.7, spawning biomass (SSB) and yield seem to have reached maximum values between 1999 and 2002, and then decreased substantially and have remained low since then. The dots in the yield plot represent the total fleet discard weights estimated in Chapter 3. These overlap with the values of estimated yield from the model. Mean fishing mortality stayed at the same high level from the start of the time series until 2000. From that point onward mean F increased to almost the double, but has steadily decreased until the most recent years. The scale of mean F is notably high, with values between 1.5 and 2.5. The fishery selectivity reaches a maximum selectivity at age 2 and then decreases at the oldest age.

Residuals checking The residual plot in Figure 5.8 does not show major patterns over years or across ages. There does not seem be any specific trend in terms of the direction of the residuals, with negative and positive residuals being equally interspersed. The magnitude of the residuals seems to be higher for surveys 2, 3 and 4, but this could be due to the fact that there is few data for these surveys compared with the catches and survey 1. Overall, there is not any clear pattern in the residuals.

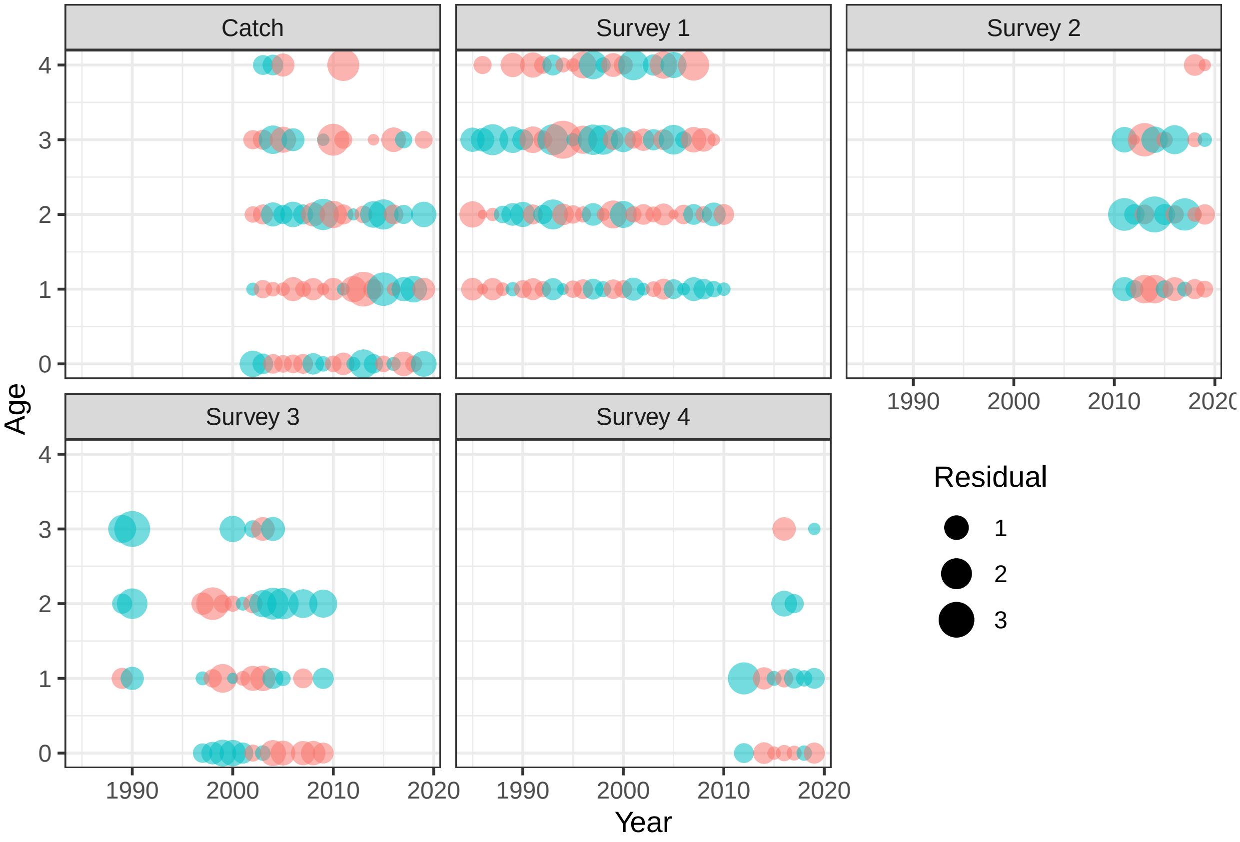

Figure 5.8: Residual bubble plots by fleet from the reference model applied to whiting data in the Clyde. Catch residuals, surveys 1 - 4 residuals. Survey 1 - Q1 survey 1985-2010; Survey 2 - Q1 survey 2011-2019; Survey 3 - Q4 survey 1985-2010; Survey 4 - Q4 survey 2011-2019. Blue = negative residuals, Red = positive residuals. Figure legend shows magnitude of residuals.

Retrospective analysis The retrospective runs for the reference model are shown in Figure 5.9. In terms of mean fishing mortality (Fbar), none of the model runs lie outside of the 95% credible interval, but there is a lot of variability in the terminal mean F values between the model runs. The estimates of mean F have the highest variability between model runs than the other quantities. For the recruitment, yield and SSB time series, excluding most recent years of data does not result in any major change of trends. Overall, although terminal F is not reliably estimated with high variability between model runs, the exclusion of up to 5 years of the most recent data does not result in major historical revisions of spawning stock biomass and recruitment.

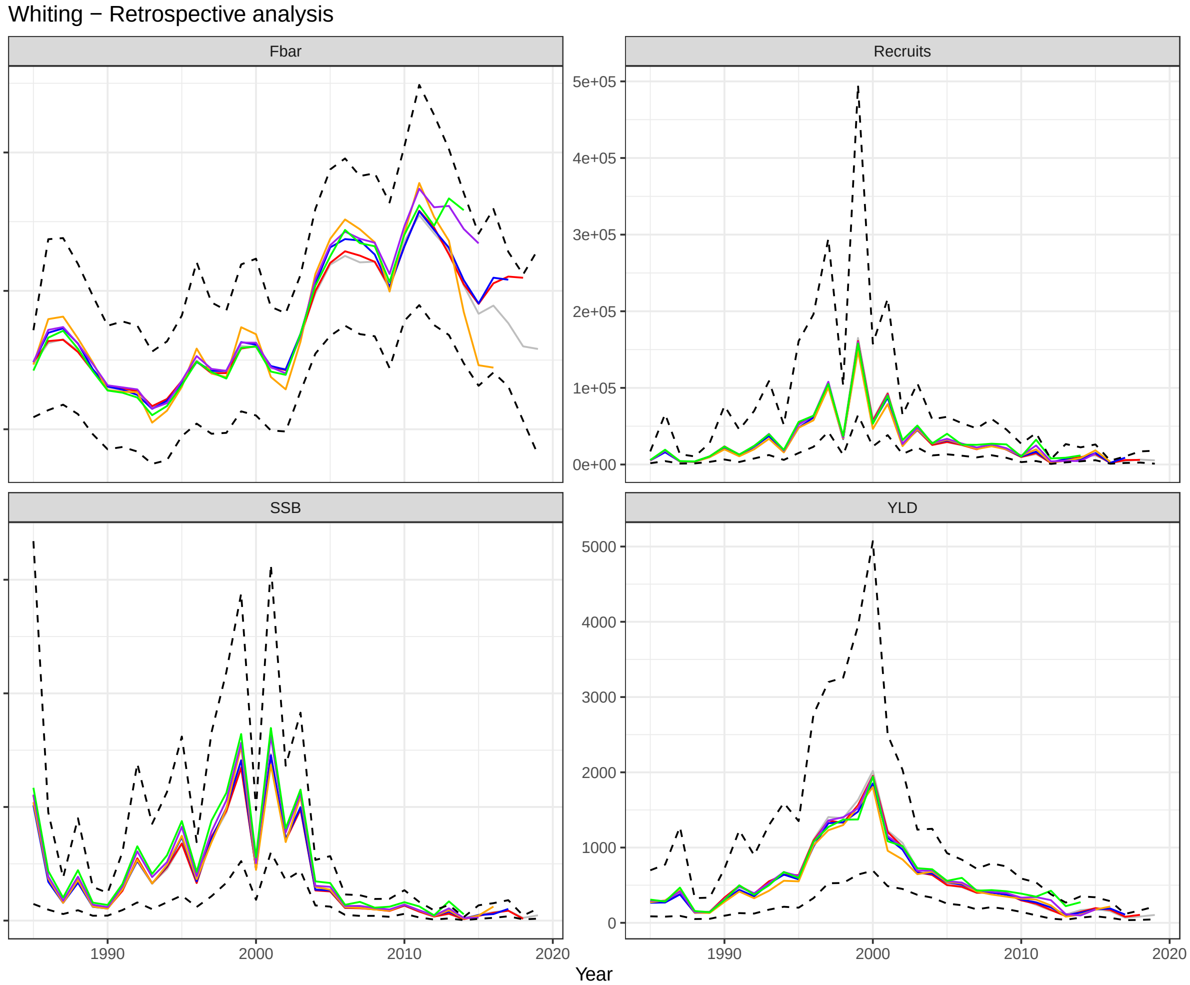

Figure 5.9: Retrospective plots for the reference model applied to whiting data in the Clyde. Dashed lines represent 0.95 credible intervals of the model run containing all years. Fbar is mean fishing mortality, recruits are in numbers of fish at age 0, SSB is spawning stock biomass (tonnes) and YLD is total catch or yield (tonnes).

Comparison with adjacent stocks Summary statistics for the Clyde whiting stock were compared with adjacent stocks (west of Scotland and Irish Sea; Figure 5.10). Mean fishing mortality in the Clyde remained at the same level until 2000, after that it increased rapidly until reaching a peak in 2011 and then decreased steadily until the most recent year. The scale of mean fishing mortality is extremely high, with values varying between 1.5 and 2.5, significantly higher than the values calculated in the ICES whiting assessment for both the Irish Sea and west of Scotland. For the west coast of Scotland, mean F is calculated at less than 0.2 for the most recent years. For the Irish Sea, mean F values are considerable higher and closer to the ones estimated in the Clyde, reaching values of 1.5 in mid-2000's and for the most recent 10 years around 0.5 and 1. All three assessments show a decrease in mean fishing mortality after 2010, although this decrease started at an earlier period for the west of Scotland and Irish Sea.

Spawning biomass in the Clyde attained maximum values between 1999 and early 2000's, and then decreased quickly to a minimum level from 2005 onward and remains low until 2019. The peak of SSB in the Clyde was not estimated for any of the adjacent stocks during that period. Since there is no catch data included in the model prior to 2002, this historical peak in SSB needs to be interpreted with caution. A decrease in biomass of whiting has also been estimated for the West of Scotland and the Irish Sea, but occurred at an earlier period, prior to 2000. The current biomass levels are low for all three stocks, although for the west of Scotland and Irish Sea it appears there is a slight increase in biomass in recent years.

Recruitment in the Clyde reached its highest peak in 1999. Before that, it appears there was very low recruitment events in the Clyde. In contrast, for the west coast and Irish Sea, recruitment starts at high values and then decreases steadily, reaching minimum values in the 2000's. There is a small peak in recruitment for the west coast that coincides with the highest peak of recruitment in the Clyde. In recent years, recruitment has increased slightly for the west coast and Irish Sea, which might be driving the small increase in spawning biomass levels. The natural mortality values used for whiting in the Clyde are slightly higher than the ICES values used for the west coast and Irish Sea.

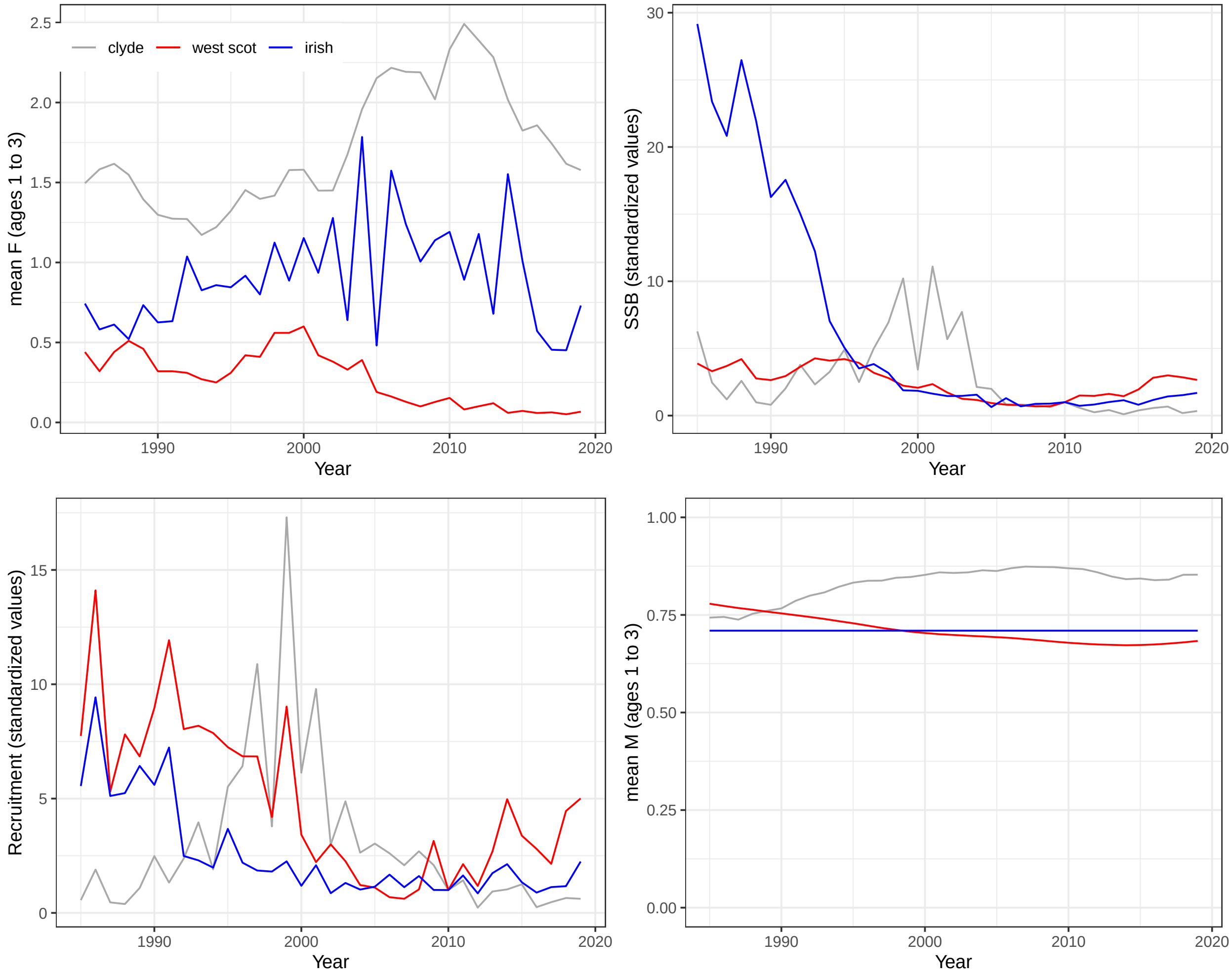

Figure 5.10: Comparison of summary statistics estimated for the Clyde whiting stock with the Irish Sea and west of Scotland areas. Mean F represents mean fishing mortality, SSB is spawning stock biomass, mean M is mean natural mortality. SSB and recruitment values were standardized to 2010 values.

5.3.3 ASM applied to Cod data

The base model was compared with each of the sensitivity scenarios in Figure 5.11 (base model represented by the grey lines). Model run A contained a reduced time series of survey data, with data from 2002 to 2019 so it would overlap with the same time period of existent catch data. Summary statistics of model run A show a high degree of similarity to the base model, particularly in terms of spawning biomass and recruitment (run A shown as the pink lines). The mean fishing mortality is lower in some years, but with a higher peak in 2013/2014. This could be due to slightly different survey selectivity patterns between model runs. Overall, the outputs of run A are similar enough to the base model to conclude that excluding the earlier part of the time series of the survey does not change significantly the model results.

For model run B, the fishery selectivity functional form was changed to a logistic function, and a different set of parameters estimated. In terms of spawning biomass, recruitment and mean fishing mortality, model run B and the base model follow similar trends (run B represented as the blue lines, Figure 5.11).

For model run C, the survey data was configured as four time series from 1985 to 2019. Spawning biomass and recruitment trends are similar to the base model, but the level of SSB has changed for the early part of the time series (run C shown as the orange lines, Figure 5.11). This might be explained by an improved model fit to the survey and catch data, particularly the most recent Q1 and Q4 surveys (surveys 2 and 4), allowing the model to capture slightly different trends in the data. The mean fishing mortality shows a higher level than the base model but only between 2009 and 2014. There was no model convergence for run D (exclusion of oldest age group), so results are omitted.

For model run E, natural mortality was derived from mean weights at age calculated specifically for Clyde cod data. Spawning biomass, recruitment and mean fishing mortality trends look similar to the base model (run E shown as the green lines, Figure 5.11), except that the level of recruitment is higher for run E.

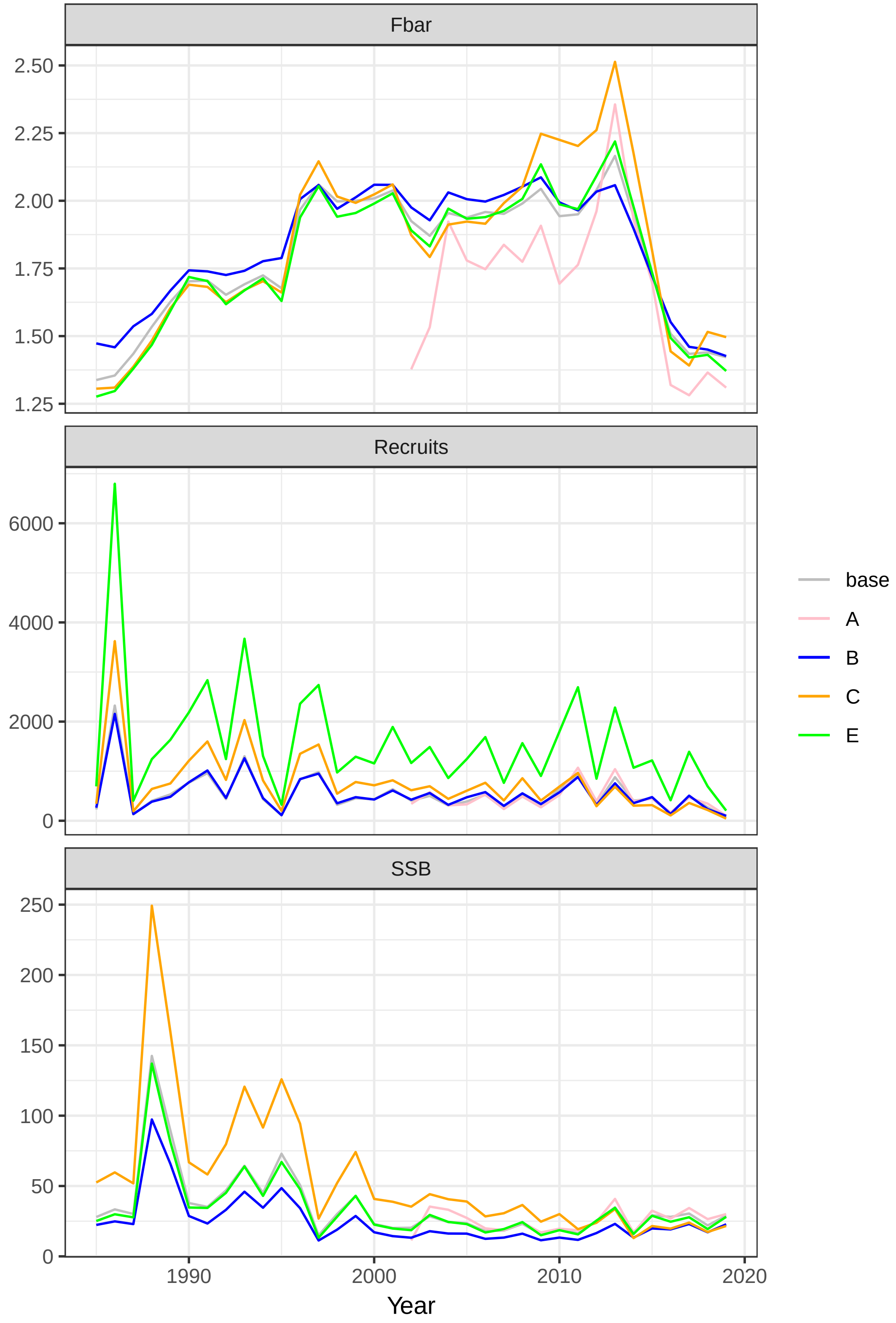

Figure 5.11: Summary statistics for the sensitivity runs applied to Clyde cod data. Fbar represents mean fishing mortality, recruits are numbers of fish at age 0, SSB is spawning stock biomass in tonnes. The sensitivity scenarios are: base model; A - reduced survey data; B - fishery selectivity as logistic curve; C - 4 surveys; E - different M assumption.

Model comparison The DIC and the effective number of parameters (pD) provide another approach for model comparison, although cannot be used to compare all the sensitivity scenarios (Table 5.9). Model run B (fishery selectivity as logistic curve) improves considerably the DIC compared with the base model, even though the outputs are identical to the base model. Model run C (4 surveys) has improved the model fit to both survey and catch data, and this is reflected in a substantial reduction of the DIC of 17 units. There was a modest increase on the number of parameters estimated for run C. After partition of the survey data into four time series as opposed to two, there were more parameters estimated overall (catchability and selectivity parameters for each survey). Model run E also results in a decrease in DIC (of 10 units) compared with the base model.

Table 5.9 - Model comparison (base model, runs B, C and E) of DIC and pD for cod data. run B - fishery selectivity as logistic curve; run C - 4 surveys; run E - different M assumption.

model	DIC	pD
base model	1346.50	46.42
run B	1331.53	45.88
run C	1329.50	50.64
run E	1336.49	45.03

Comparing all sensitivity scenarios, there are some changes to the base model that are worth considering for deciding on a final model configuration. Changing the assumption on the functional form of fishery selectivity provides an improvement to the model fit (run B). However, to keep the same model consistency across species, it was decided to still keep a gamma curve for the final model. The model run with survey data included as four separate time series (run C) improves substantially the model fit, so this configuration will be retained. Model run D did not converge (model could not reach a solution), so 5 age groups will be kept. Calculating the natural mortality from mean weights at age specific to the Clyde cod data (run E) is a more realistic assumption than relying on values for other areas.

Reference model - Final model configuration The final model, named the reference model, had the following configuration: fishery selectivity as a gamma curve; survey data included as four time series; 5 age groups (0 to 4 year old fish); two separate sets of values for the smoothed mean weights at age, weights from the Q1 survey to estimate SSB (start of the year) and annual yield, and average weights from Q1 and Q4 surveys to estimate natural mortality. The smoothed mean weights at age from the Q1 survey seem to be reasonable for calculating SSB (at the start of the year) and yield, contrary to the results for haddock and whiting. This is because there are very few cod data to calculate the mean weights for the Q4 survey, and consequently a large discrepancy between the weights from the Q1 survey and from an average of the Q1 and Q4 surveys (see Figure 2.17 in Chapter 2). Another reason is that the yield values estimated using the Q1 survey mean weights are in agreement with the discard estimates from Chapter 3 (see Yield plot in Figure 5.12). For similar reasons discussed for haddock, mean weights from Q1 and Q4 surveys were used to calculate M from the Lorenzen equation. These values are intermediate between the those used in the base model and run E (Figure D.14 in Appendix D) which will be reflected in the estimates F and SSB accordingly.

The summary statistics, fishery and survey selectivity curves from the reference model are shown in Figure 5.12. The reference model can be directly compared with the base model in terms of DIC and pD, and provides a substantial reduction of the DIC (and a slight increase in the number of estimated parameters - Table 5.10).

Table 5.10 - Model comparison (base model and reference model) of DIC and pD for cod data.

model	DIC	pD
base model	1346.50	46.42
reference model	1327.51	50.42

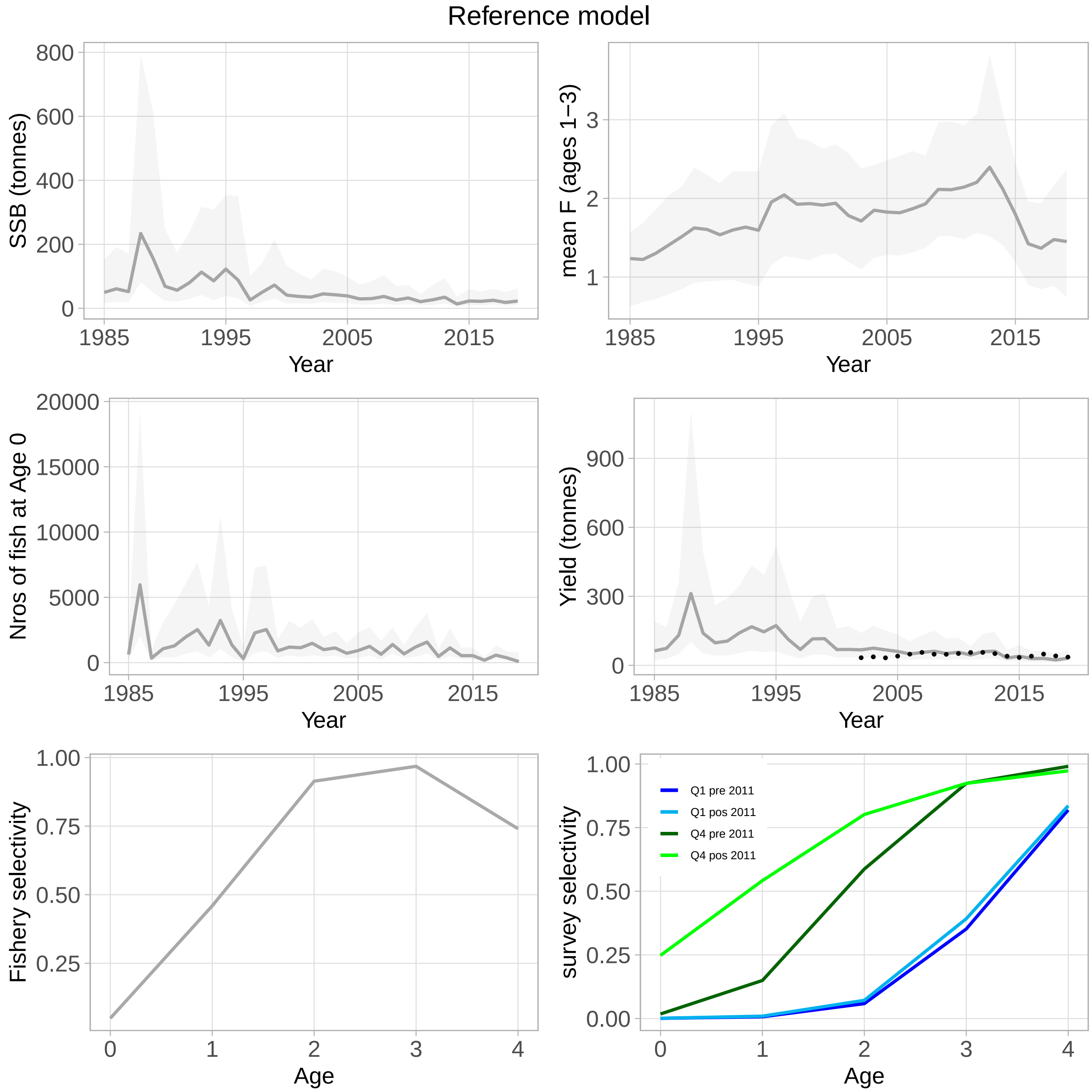

Figure 5.12: Summary statistics and selectivity curves for the reference model applied to Clyde cod data. Dots represent cod discard estimates from Chapter 3, shaded areas represent 0.95 credible intervals.

Looking at specific patterns over time in Figure 5.12, the model estimates a decreasing trend in SSB and yield from 1988 until early 2000's, and they remain at low levels. Mean fishing mortality fluctuated for the whole time series at very high values, with abrupt increases between 1985 and 1997. After that, mean F remained at the same level until 2013, where mean F reduced to the same level of low values as the start of the time series. The dots in the yield plot represent the total fleet discard weights estimated in Chapter 3. These overlap with the values of estimated yield from the model. The selectivity curve for the Q4 pre-2011 survey show a logistic curve, while both of the Q1 selectivity curves do not quite reach an asymptote for the maximum age observed. As a result, the age at 50% retention was estimated between age 1 and age 2 for the Q4 pre-2011 survey, and between age 3 and age 4 for the Q1 surveys. The fishery selectivity reaches a maximum selection at age 3, and then declines slightly at age 4.

Residuals checking The residual plot in Figure 5.13 do not show major patterns over years or across ages. For the catch residuals, there are more positive residuals at age 1, but these are still interspersed with some negative residuals. The magnitude of the residuals seems to be higher for survey 4, but this could be due to the fact that there is few data for this survey compared with the catches and survey 1.

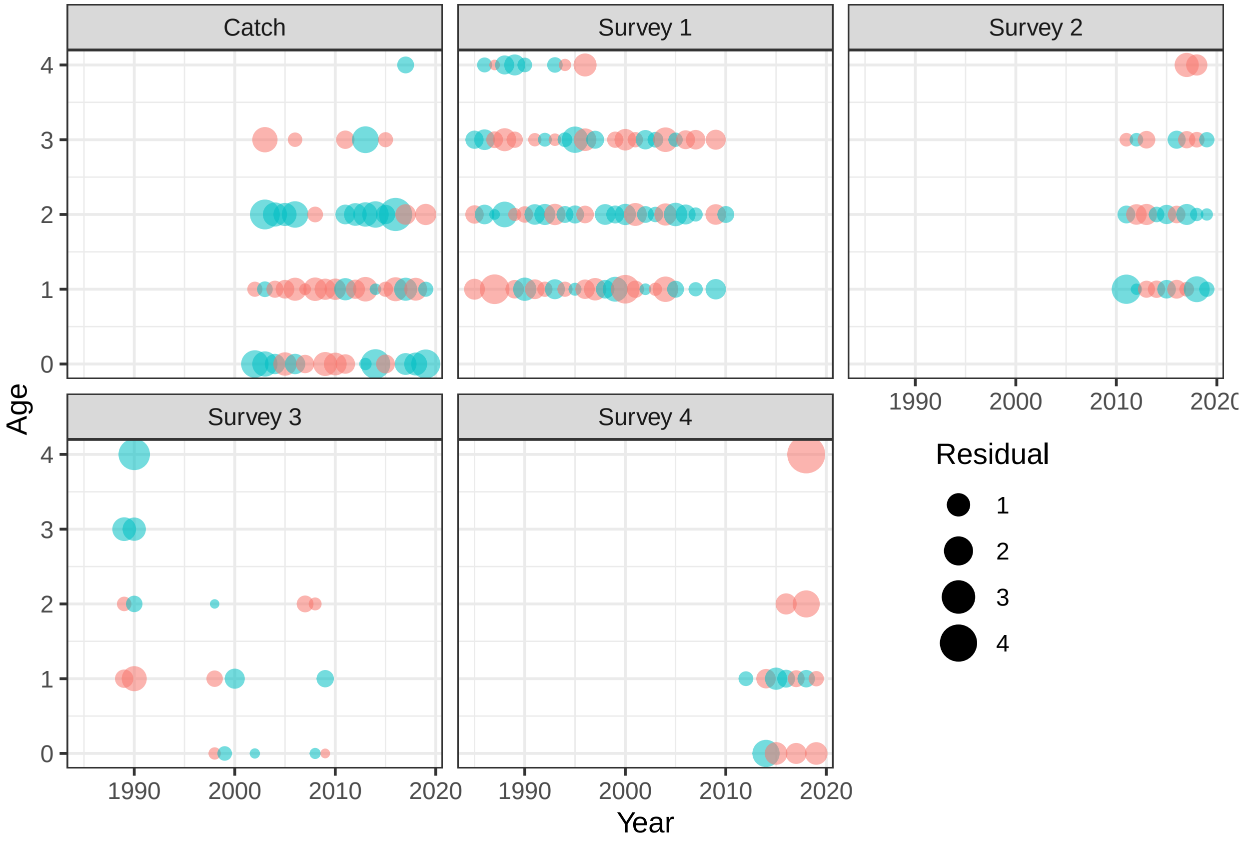

Figure 5.13: Residual bubble plots by fleet from the reference model applied to cod data in the Clyde. Catch residuals, surveys 1 - 4 residuals. Survey 1 - Q1 survey 1985-2010; Survey 2 - Q1 survey 2011-2019; Survey 3 - Q4 survey 1985-2010; Survey 4 - Q4 survey 2011-2019. Blue = negative residuals, Red = positive residuals. Figure legend shows magnitude of residuals.

Retrospective analysis The retrospective runs for the reference model are shown in Figure 5.14. In terms of mean fishing mortality (Fbar), none of the model runs lies outside of the 95% credible interval, but there is a lot of variability in the terminal F values between model runs. There seems to be a level shift consisting of a downward revision of mean F when excluding 5 years of data (green line in the plot). This can be an indication of model misspecification. Nevertheless, all the retrospective runs show the same trend in fishing mortality. For the recruitment, yield and SSB time series, excluding previous years of data does not result in any major change of trends. Overall, the exclusion of up to 5 years of the most recent data does not result in major historical revisions to spawning biomass or recruitment, but it does affect the level of mean fishing mortality, possibly indicating a model weakness.

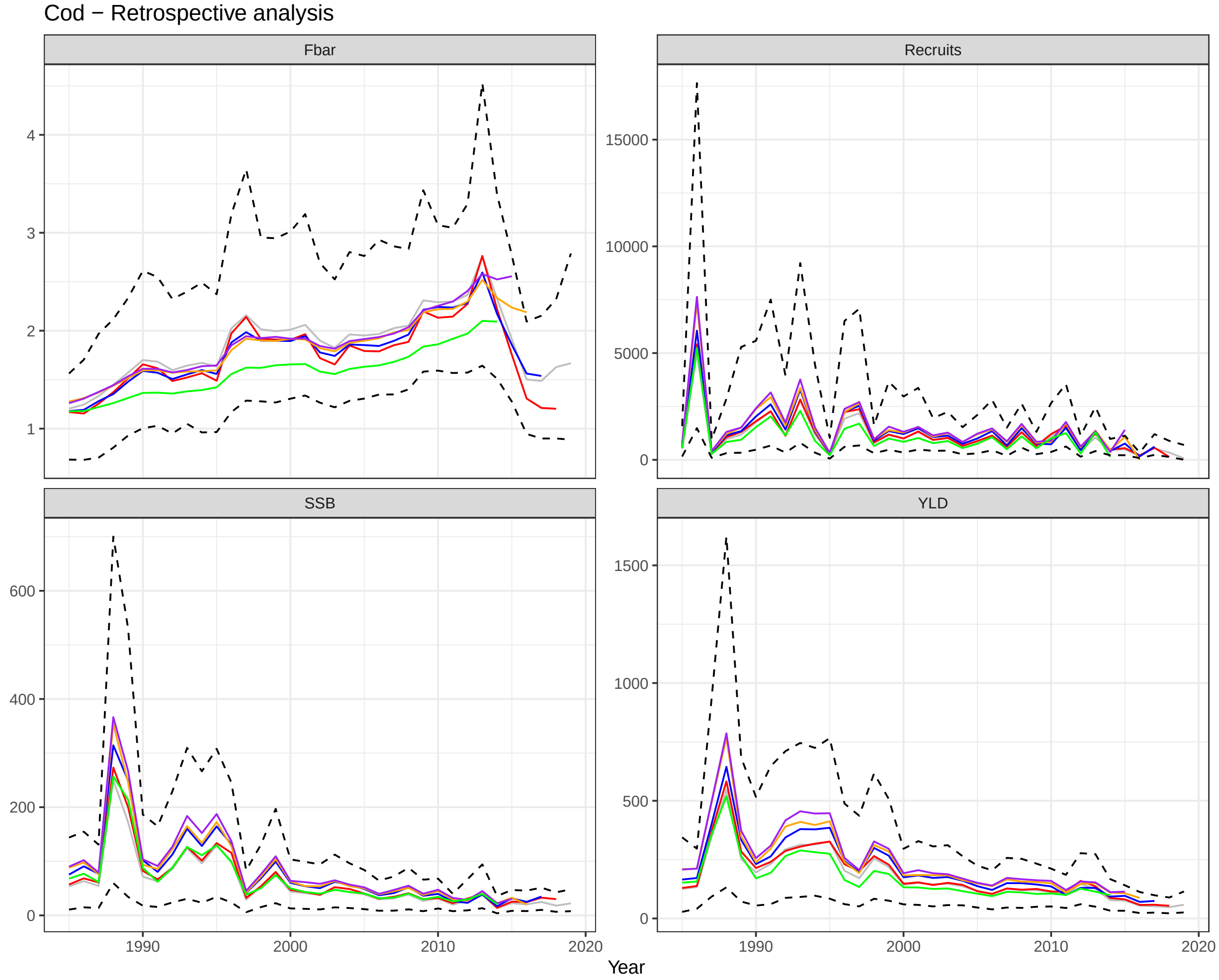

Figure 5.14: Retrospective plots for the reference model applied to cod data in the Clyde. Dashed lines represent 0.95 credible intervals of the model run containing all years. Fbar is mean fishing mortality, recruits are in numbers of fish at age 0, SSB is spawning stock biomass (tonnes) and YLD is total catch or yield (tonnes).

Comparison with adjacent stocks Summary statistics for the Clyde cod stock were compared with adjacent stocks (west of Scotland and Irish Sea; Figure 5.15). Mean fishing mortality in the Clyde seems to have increased over the time series until reaching a maximum after 2010, and quickly decreasing to the same level as of 1985. This is an unexpected pattern, as drastic changes in fishing mortality from one year to the next are less likely to occur, so care must be taken when interpreting apparent changes in mean F. The overall level of fishing mortality in the Clyde is considerably higher than the values from the ICES assessments for adjacent stocks. In the Clyde, mean F is estimated to vary between 1.5 and 2.5 across the time series. For the most recent years, mean F is estimated around 0.4 for the west of Scotland stock, and at less than 0.2 for the Irish Sea stock, while for the Clyde stock is situated around 1.5.

Spawning stock biomass of cod in the Clyde steadily decreased since the 1990's, and is now at historically low levels (biomass less than 50 tonnes per year). The ICES assessments for the west of Scotland and Irish Sea provide similar results, with spawning biomass decreasing sharply from 1985 to early 2000's (ICES, 2018; ICES, 2021a). However, the Irish Sea assessment estimates a rapid increase of biomass from 2013 onward, which contrasts with the flat trend of low biomass in the Clyde. The west coast assessment also estimates a modest increase in biomass after 2015, but it decreases shortly afterwards. Recruitment has decreased substantially for the west coast from 1985 to 2000. After that, recruitment has remained at low levels for all the three stocks. The natural mortality values in the reference model used for cod in the Clyde are similar to the ICES values used for the west of Scotland and Irish Sea.

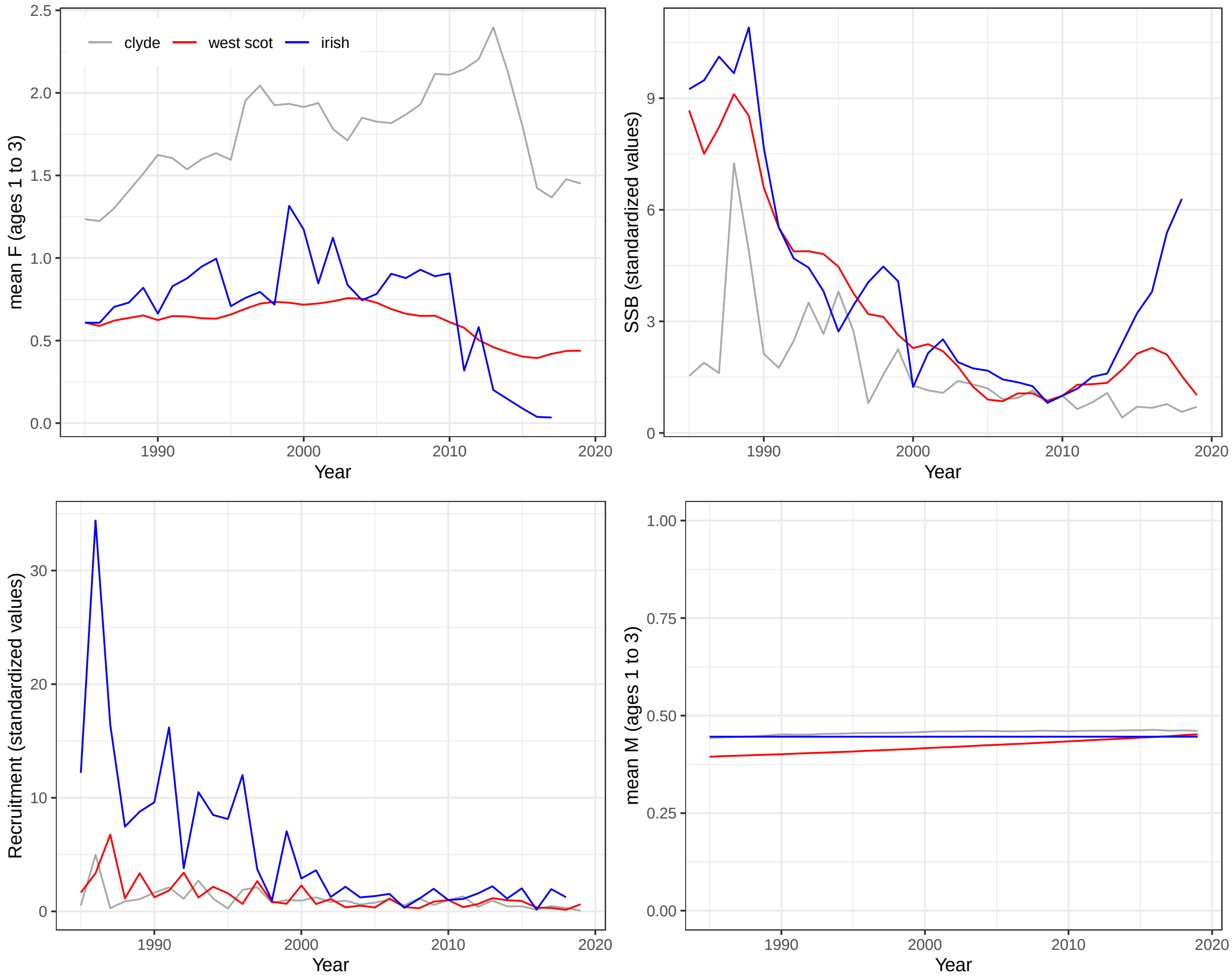

Figure 5.15: Comparison of summary statistics estimated for the Clyde cod stock with the Irish Sea and west of Scotland areas. Mean F represents mean fishing mortality, SSB is spawning stock biomass, mean M is mean natural mortality. SSB and recruitment values were standardized to 2010 values.

5.4 Discussion

A Bayesian age-structured population model was developed specifically for the Clyde stocks of cod, haddock and whiting. The model appears to fit to both survey and catch for each species reasonably well (Figures D.2 - D.3, D.8 - D.9 and D.12 - D.13 in Appendix D) and successfully estimates the parameters of interest, although there were few observations for the Q4 survey to fit the model, particularly for cod. It allows for the calculation of quantities of interest such as fishing mortality and spawning stock biomass. The sensitivity analysis led to a model configuration that best suited the characteristics of the data for each species, and tested the robustness of the model to alternative assumptions. These were changing the assumption on fishery selectivity, omitting survey data, including/excluding oldest age groups, changing the configuration of the survey data, and changing the values of natural mortality to reflect realistic values calculated for the Clyde species. The DIC (and effective number of parameters) provided a direct measure of model comparison, while the residuals and retrospective analysis for the reference model allowed detection of problems in the model fit and major bias in the model outputs.

The final model configuration was similar across the three species, only small differences were adjusted for each. There were no survey selectivity parameters estimated for the whiting data, because the model could not distinguish a different selectivity value for each age class. The smoothed mean weights at age (estimated as an average from the Q1 and Q4 surveys) were used to calculate yield for haddock and whiting. However, for cod, the smoothed mean weights estimated only from the Q1 survey seemed to be adequate to estimate yield. This is because there are few observations for the Q4 survey for cod, and consequently a large discrepancy between mean weights of the Q1 and Q4 surveys. From the plots shown in Chapter 2 (Figures 2.16 - 2.18), it can be seen that cod also reaches larger mean weights at age than haddock and whiting. Hunter et al. (2016) estimated age-length data for gadoid species in the Clyde, which is relevant considering length and weight are related measures to size. Their results showed that for the youngest age group (age 1), the mean length is similar for the three species, around 20 cm. However, as fish grow, cod reaches considerably larger mean lengths at age than haddock and whiting. At age 3, mean length for whiting varies between 20 to 30 cm, for haddock between 30-40 cm, and cod measures around 60 cm. This means that over the course of a year, cod grows quicker than the other species, so the mean weight at age from the Q1 survey will be lower than the mean weights calculated at the end of the year when the Q4 survey takes place.

Comparing the sensitivity analysis results between species, some changes in assumptions affected to a greater extent the model outputs of certain species as opposed to others. The impact on results of model run A, when omitting historic values of survey data, was similar for all three species. Mean fishing mortality across ages 1 to 3 decreased slightly, and survey and fishery selectivity curves have changed to some extent but not with the same direction of change across species. When changing the assumption on fishery selectivity from a gamma to a logistic functional form (model run B), there was a upward shift of mean fishing mortality for haddock, but this was not observed for whiting or cod. This level shift for haddock might be explained by a change of the estimated survey selectivity curves, even though the model was not able to properly estimate these parameters. For model run B, there was a substantial improvement in DIC for cod and a slight improvement for whiting, but the same was not observed for haddock. However, the reference model (that included a gamma curve) still had a lower DIC than model run B. In order maintain the same fishery model across stocks, it was decided to keep the fishery selectivity assumption as a gamma curve for the reference model for all three species. Dome shaped fishery selectivity is expected even when gear selectivity is asymptotic (Sampson and Scott, 2011) and would therefore seem appropriate in this case. Including the survey data as 4 separate time series (model run C) also led to differences in spawning biomass, recruitment and mean fishing mortality for all species. While for haddock there was a change in the level of mean F but the trend remained exactly the same, for whiting and cod there appears to be some variation of the mean fishing mortality trends across years. The selectivity parameters for the Q4 post-2011 survey were poorly estimated for haddock due to the lack of data points (Figure D.2 in Appendix D). Nevertheless, there was a considerable improvement of the model fit given by the DIC and pD criteria for all species. When excluding the oldest age group (age 4) and applying the model to cod data (model run D), the model was not able to converge, likely because there was not enough data to allow the model to find a solution. Including one extra age group for haddock resulted in a large shift of mean fishing mortality to higher values, probably due to changes in both survey and fishery selectivity curves. On the other hand, adding an older age group to the whiting data did not result in any major change. The reference models for the three stocks included 5 age groups (0- to 4-year-olds). Natural mortality derived from mean weights at age estimated specifically for the Clyde stocks provided a more realistic approach than relying on values calculated for adjacent areas, reflecting the decreases of mean weights-at-age and consequent increases of natural mortality at age over time for the Clyde haddock and whiting stocks. The changes in the model outputs depended on the values of natural mortality used, and improved the model fit for cod and haddock, but not for whiting (DIC value for whiting increased 0.45 units).

The sensitivity analysis helped identify a reference model configuration. The results of the reference model were compared to the most adjacent stocks of the west of Scotland and Irish Sea for each species. For haddock, there has been a recent increase of spawning biomass in the Clyde, likely derived from strong recruitment events. This increase in biomass is also observed for the Irish Sea and, to some extent, west of Scotland, and a consequent decrease in fishing mortality for all three populations, although the absolute level of fishing mortality is different between these. There appears to be some connectivity between the three areas, providing an indication that the Clyde haddock population might not be fully isolated from adjacent stocks. The wider dispersal movements across regions are likely to be driven by the early life stages (larvae and juveniles), when there appears to be a high degree of connectivity between areas such as the North Sea, Skagerrak and the west of Scotland (ICES, 2014). In contrast, most adult haddock show little dispersion movements, and adopt small home ranges, with the example of the Clyde where most adult haddock have the otolith composition signal of the local juvenile area (ICES, 2014). Even if the haddock population would be differentiated into multiple subpopulations in the Clyde, Irish Sea and west of Scotland, it is likely that all three subpopulations will suffer similar fishing pressures and environmental conditions and might respond with similar trends of spawning biomass and recruitment across areas.

For whiting, there are no obvious similarities in terms of spawning biomass and recruitment trends between the Clyde and adjacent stocks. For all three whiting assessments, fishing mortality seems to have decreased for the last 10 years, and spawning biomass remains at low levels.

For cod, spawning biomass has decreased substantially from 1985 to low levels around 2010, and this trend was registered for all the three cod assessment areas. However, after 2010, it seems that biomass has increased in the Irish Sea and west of Scotland, while for the Clyde it continued to decrease to a historical minimum level. This provides evidence of distinct patterns of cod spawning biomass in the Clyde. It seems that fishing mortality has decreased in the Clyde after 2010, which is also registered for the 6a and 7a areas.

One striking outcome observed for all the three species is the higher values of mean fishing mortality estimated for the Clyde compared to the west of Scotland and Irish Sea. There could be several reasons why the level of fishing mortality estimated for haddock, cod and whiting in the Clyde is higher than for other areas. One could be related with the values of natural mortality used in the assessment. Natural mortality includes sources of mortality like predation, disease and migratory movements (emigration does not necessarily represent mortality, but a part of the population that disappears from the stock area). The model used here only tracks the total mortality of the cohorts, so if the values of natural mortality are underestimated, then this could result in an overestimation of the fishing mortality component. In contrast, higher assumed natural mortality values will result in a lower level of fishing mortality and consequently an increased stock biomass. In the west coast of Scotland, recent increases in the seal population that predate on gadoid species can be an important component of the total stock mortality, particularly for cod (Cook et al., 2015). However, this source of mortality is hard to quantify, and there is no information available at the Clyde level. Besides, the natural mortality values were estimated using a similar approach as used by ICES and are in close agreement with these, at least for whiting and cod. Natural mortality for the 6a haddock stock until 2013 was considered as a conventional value of 0.2 that does not change with age or time, and not estimated from mean weights at age, thus explaining the different values of natural mortality between the three haddock stocks. Another hypothesis for an overestimation of fishing mortality can be migratory patterns to areas outside the Firth of Clyde. Both the Nephrops fishery and the survey trawl caught mostly young whiting in the Clyde (Chapter 2). Burns et al. (2020) has shown that the Clyde might act as a nursery ground for whiting, with adults moving away to further offshore waters after a certain age. It is also known that haddock shows wide dispersion movements and connectivity across large areas, and haddock in the Clyde seems to have appeared after 1990 derived from recruitment classes of adjacent stocks, since the extremely low levels of spawning biomass between 1985 and 1990 could not have originated the high recruitment values estimated at this time (Dickey-Collas et al., 2003). On the other hand, there is substantial research indicating that the Clyde cod stock is an isolated unit, from tag-recapture data, genetic and otolith markers, and there is a need to assess the cod population in the Clyde independently of the wider west coast as per the cod stock identification report from ICES (ICES, 2022c). As such, it does not seem plausible that the emigration patterns could explain the apparent high level of fishing mortality, at least for cod, but also for whiting and haddock since it was used the same methodology for assessing all species.

The most likely explanation for the high level of estimated fishing mortality is that the Clyde, representing a small area compared with the other assessed areas, concentrates a high amount of fishing effort that leads to values of fishing mortality that are much higher than in other areas. A comparison of fishing intensity calculated in days per km^2 between the Firth of Clyde, the West coast of Scotland and the Irish Sea shows that the Clyde has considerably higher values (Figure 5.16). Data on fishing effort was subset for UK bottom trawlers over 10 m length, and then divided by total area. Since fishing mortality is related to fishing effort, the more intensive effort in the Clyde might explain the higher level of mean F. The values for the Irish Sea are intermediate between the Clyde and west coast values, which matches exactly the difference in fishing mortality level estimated for the whiting stocks (Figure 5.10).

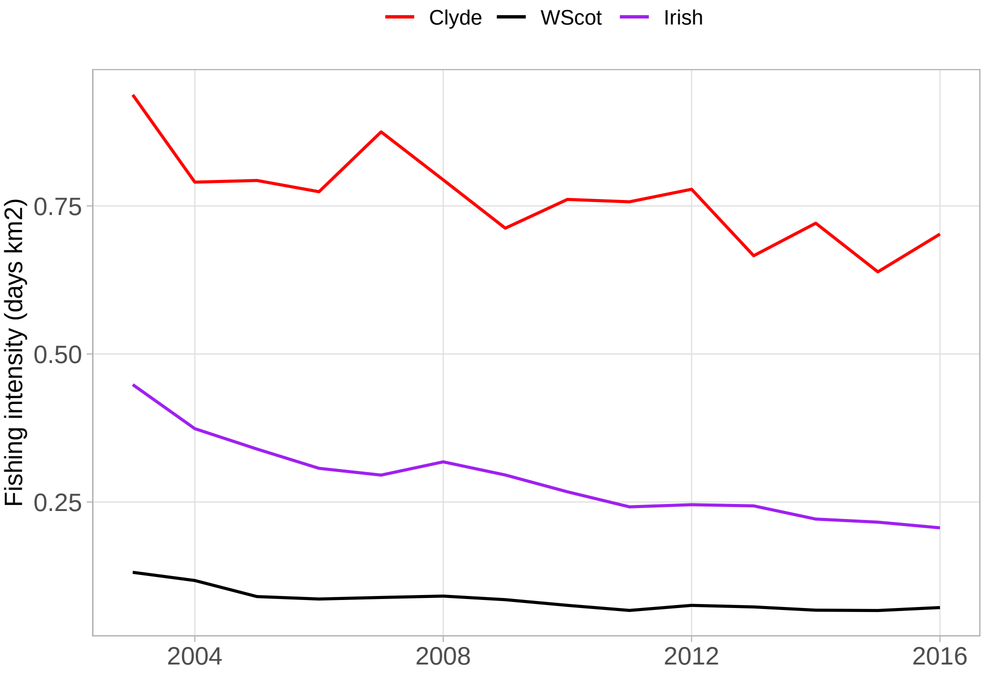

Figure 5.16: Annual fishing intensity of UK bottom trawling vessels over 10 m from 2003 and 2016. WScot represents west of Scotland, Irish represents the Irish Sea. Data extracted from STECF (Zanzi and Holmes, 2017).

5.5 Conclusions

Overall, the ASM provides a picture of a substantial decrease in biomass for whiting and cod in the Clyde to historical minimum levels, and values of mean fishing mortality remaining at very high values. The haddock population shows another pattern, with a recent increase of spawning biomass derived from strong recruitment events and a decrease in fishing mortality, which matches the pattern registered for the west of Scotland and the Irish Sea estimated by ICES. Fishing mortality for all the three stocks in the Clyde is remarkably higher than fishing mortality estimated by ICES for adjacent areas, and this is likely to be explained by the relatively higher fishing effort intensity occurring in the Clyde.

CHAPTER 6 - Surplus Production Model

6.1 Introduction

Surplus production models group the overall effects of recruitment, growth and mortality (all aspects related with production) into a single pool of undifferentiated biomass (Haddon, 2021). They do not include biological characteristics such as age or length-structure, maturity or natural mortality. This category of stock assessment models tracks the biomass trajectory of a population over time, depending on a few parameters such as the intrinsic growth rate of the population, the carrying capacity (maximum biomass that the population's environment can sustain) and a catchability coefficient related to fishing mortality. A stock-recruitment relationship is implicit in the stock dynamics of surplus production models.

There are some difficulties in fitting surplus production models. There is often severe parameter confounding between population growth rate, carrying capacity and the catchability coefficient. To alleviate this issue, the data needs to have sufficient contrast, meaning there needs to be data available for a range of stock abundance levels and fishing intensity levels, which is not always the case, and especially if the population shows only a declining trend (Hilborn and Walters, 1992).

The Schaefer model (Schaefer, 1954) used in this work is a simple logistic growth model, with an added term representing the catches. The catches are assumed to be proportional to effort and stock size (Quinn and Deriso, 1999). The main feature of the Schaefer model is a symmetric relationship between surplus production and population biomass. Surplus production is zero at a biomass of zero, and at a biomass corresponding to maximum carrying capacity (Hilborn and Walters, 1992). Also, a population is optimally exploited if it is fished to remain at the biomass level that results in maximum sustainable yield (MSY), that can be directly calculated from the parameters of the Schaefer model.

State-space Schaefer models with stochastic population dynamics have been in use for a number of years. These assume that biomass and fishing mortality are state processes, which are observed indirectly through survey indices and/or commercial catches sampled with or without error. The majority of the model developments adopted a discrete-time form that is able to estimate simultaneously process and observation error, using a Bayesian approach (Meyer and Millar, 1999) or a frequentist approach (Punt, 2003). More recently, a continuous time version (Surplus Production model in Continuous Time, SPiCT; Pedersen and Berg, 2017) accommodates irregularly sampled data without the need for catch and index observations to match temporally. Another example is a Bayesian surplus production model that accounts for increases in fishing power over time (Cook et al., 2021).

A Bayesian surplus production model in the form of a Schaefer model was explored in this Chapter as an alternative model to the age-structured model (ASM) developed and applied in Chapters 4 and 5. Due to its few data requirements, the Schaefer model allows the inclusion of historical catch data for which age compositions could not be extracted and thus not used in the ASM. Although the ASM seems to provide reliable estimates of fishing mortality and stock biomass, it is good practice not to rely on a single best model that may imply a narrower range of uncertainty about the assessment than is actually the case (Cook, 2019a; Patterson et al., 2001). Using an alternative model structure with completely distinct population dynamics allows the comparison of trends and scale of critical stock parameters and to provide a robust assessment of the Clyde stocks.

In this chapter, a Schaefer surplus production model is developed and applied to the Clyde demersal fish stocks (haddock, whiting and cod). The Schaefer model equations are described in detail, and the model is tested with example data and compared with the standard surplus production model (SPiCT) used within the ICES framework. Model sensitivity analysis was conducted by including different subsets of data and changing model assumptions, and the stock biomass estimated from the Schaefer model was also compared with the results from Chapter 5.

6.2 Methods

6.2.1 Model description

A common form of a surplus production model can be written that expresses the population dynamics in terms of unfished biomass or initial biomass available in the time series (B_0) and an intrinsic growth rate of biomass (r). The Schaefer model described here is named Schaefer delay difference model (SDDM), since it contains difference equations with a one-year time step in the biomass dynamics. The stock biomass (B_y) is projected forward each year y using the following equation (Hilborn and Walters, 1992):

$$B_{y+1} = \left(B_y + rB_y\left(1 - \frac{B_y}{B_0}\right) - C_y\right)\ e^{(\varepsilon_y)} \tag{6.1}$$

where C_y is the total catch per year across fishing fleets. The exponential term is a random process error to account for random recruitment effects or biomass changes:

$$\varepsilon_y \sim normal(0, \sigma_B) \tag{6.2}$$

The catch is assumed to be proportional to the stock biomass through an annual fishing mortality, F_y such that

$$C_y = B_y\ F_y \tag{6.3}$$

which allows the calculation of the fitted catches from the model. Fishing mortality follows a random walk with standard deviation σ_f:

$$F_y \sim lognormal(log(F_{y-1}), \sigma_f) \tag{6.4}$$

We assume the survey index is proportional to the fish biomass in the sea, with a survey specific catchability parameter q_k:

$$U_{y,k} = q_k B_y \tag{6.5}$$

The model is parameterised in terms of F_{msy} where $F_{msy} = \frac{r}{2}$ and in terms of carrying capacity, B_0 or the unfished equilibrium stock size.

Error distributions The observed survey indices are assumed to be lognormally distributed with survey-specific standard deviation

$$U'_{y,k} \sim lognormal(log(U_{y,k}), \sigma_k) \tag{6.6}$$

The observed catch is observed with lognormal error:

$$C'_y \sim lognormal(log(C_y), \sigma_C) \tag{6.7}$$

6.2.2 Data

Biomass-based indices of abundance from the survey data and commercial fisheries data were used for the three main species in the Clyde. The calculation of the survey index based on total weights per hour of fishing effort has been previously explained in Chapter 2. Four time series of survey indices were used: Q1 survey between 1985-2010; Q1 survey between 2011-2019; Q4 survey between 1985-2010; and Q4 survey between 2011-2019. Discarded weight for the observer trips between 1990-2019 for the Nephrops fleet in the Clyde was extracted (as explained in Chapter 2), together with the fish landings by the Nephrops fleet. During the period between 1990 and 2001, there were also other fishing fleets operating in the Clyde, such as light trawls and Danish seine nets targeting fish species (Hislop, 1986). For this historical period, we can consider two categories of fishing vessels and gears: the Nephrops fleet, operating nets with mesh size between 70 and 100 mm (TR2 fleet); and the demersal fish fleet, which targeted demersal fish with nets larger than 100 mm (TR1 fleet). The fish landings for the TR1 fleet were extracted from Marine Scotland Science data bases. However, there were no observer trips sampling the discarded portion. In order to account for this, a discard rate previously estimated by Fernandes et al. (2011) for the whole of the West of Scotland was used to approximate the discarded proportions of the TR1 fleet in the Clyde. The fish landings and discards were first calculated separately for each fleet in the Clyde (TR1 and TR2 fleets), then the data was grouped into a single category of total catches (summed across fleets). The four time series of survey and catch data available are listed in Table 6.1 with the years used.

Table 6.1 - Research vessel surveys and catch data used in the model.

Data source	Years used
Survey 1 (Q1)	2011-2019
Survey 2 (Q1)	1985-2010
Survey 3 (Q4)	2011-2019
Survey 4 (Q4)	1985-2010
Total catch	1990-2019

6.2.3 Model testing

The SDDM was tested on an example data set in order to compare the outputs with the SPiCT model (Pedersen and Berg, 2017). SPiCT is a stochastic surplus production model in continuous time, which has similar base assumptions as the SDDM developed here. The example data is a data set of South Atlantic albacore (*Thunnus alalunga*), previously published in Polacheck et al. (1993). It contains 23 years of one time series of catch weights and one time series of CPUE (in kg per 100 hooks), from 1967 to 1989. This example data set is available within the SPiCT R package. Both the SPiCT and SDDM models were fitted to the same example data.

By fitting the SDDM to the same example data, the SPiCT model that is routinely used within the ICES assessment framework should provide a conventional model to be used as means of comparison to confirm how realistic are the outputs from the SDDM. If the estimated trends of fishing mortality and stock biomass are in reasonable agreement between the two models, this indicates that the SDDM is likely to perform as well as SPiCT on the Clyde stocks. The main difference between the SDDM and SPiCT models is that SPiCT assumes that the catches can be taken throughout the year, while the SDDM assumes that catches need to be taken at the start of the year.

6.2.4 Model configuration

Parameters were estimated by fitting the model to the catch and survey time series using the R package "RStan" (Stan Development Team, 2023). Three MCMC chains were run with a minimum of 50 000 iterations, a burn in of 25 000 and a thinning rate of 50. Model convergence was assessed with traceplots and calculation of the Rhat value. Histograms of the marginal posterior distributions of F_{msy} and B_0 are shown for the base model as well as histograms of the Rhat distribution. The joint distribution of F_{msy} and B_0 is represented as contour plots.

Priors on the parameters and their description are shown in Table 6.2. A weakly informative square root prior on B_0 was chosen so that it could be estimated without excessive bias (Cook et al., 2021). F_{msy} is defined over the range of values between 0 and 1. A beta prior, $Beta(2,3)$, was applied to F_{msy}, which has a mode of approximately 0.3 that corresponds to the median of a meta-analysis of F_{msy} for a range of North Sea stocks (Sparholt et al., 2020). Other priors are simple uniform distributions. Preliminary model runs for whiting indicated that the posterior distribution of B_0 was hitting the upper bound of the prior, so it was expanded to $Uniform(1, 500)$.

Table 6.2 - Prior distributions on the Schaefer model parameters.

Parameter	Prior	Description
$\sqrt{B_0}$	$Uniform(1, 100)$	Carrying capacity or virgin biomass (square root scale)
F_{msy}	$Beta(2, 3)$	Fishing mortality at MSY
B_1	$Uniform(0.01, 3 \cdot maxcatch)$	Biomass in the first year
q_k	$Uniform(0.001, 20)$	Survey catchability coefficient for fleet k
σ_{surv}	$Uniform(0, 10)$	Standard deviation of observation errors in surveys
σ_C	$Uniform(0, 10)$	Standard deviation of observation errors in catch data
σ_f	$Uniform(0, 1)$	Standard deviation of fishing effort process

Parameter	Prior	Description
σ_B	$Uniform(0,1)$	Standard deviation of biomass process error
$F_{y=1}$	$Uniform(0,4)$	Fishing mortality in the first year

6.2.5 Sensitivity analysis

For each species, alternative model configurations were tested to understand how including different subsets of data and changing model assumptions affected the outputs in terms of biomass estimates and fishing mortality. This provides a way to check model robustness and consistency, and the estimated biomass can be compared with the stock biomass calculated previously in Chapter 5.

The SDDM was tested with 4 different model configurations, but with the same structural model and base assumptions (Table 6.3). Run 1 consisted of running the model including only the survey data, without using the catch data. No catch observation error was included in the priors. Model run 2 included both the survey data and the catch data, with the catches between 1990 and 2019. Model run 3 included both the survey and catch data, except that the catch time series was reduced to data between 2002 and 2019, in order to overlap with the same time series of catch data included in the age-structured model (ASM) from Chapter 5. Since this run is the most similar to the reference ASM, it was chosen as the "base model". Model run 4 included survey and catch data, but the catches were included as known values (model not fitted to the catches). This means that fishing mortality was estimated directly from the catches and the estimated biomass as $F_y = C_y/B_y$. No process error on F was included nor priors on F or catch observation error.

Table 6.3 - Sensitivity model runs used for SDDM for all three species.

Model	Data input
Run 1	Survey data only
Run 2	Survey and catches (catches between 1990-2019)
Base model	Survey and catches (catches between 2002-2019)

Model	Data input
Run 4	Survey and fixed catches (catches between 2002-2019)

6.3 Results

6.3.1 Model testing

Results of testing the model with example data to compare how the SDDM performed in relation to the SPiCT model are shown in Figure 6.1. The SDDM produced similar trends in stock biomass and fishing mortality as the SPiCT model. Table 6.4 compared the values of B_0 and F_{msy} estimated for both models. The SDDM gives a similar value of B_0 as SPiCT, but with very wide confidence intervals. This can be explained by the highly skewed distribution of B_0 shown in Figure E.1 in Appendix E. In the SDDM the prior restricts the estimate of F_{msy} and gives a lower value of F_{msy} compared with the SPiCT model, but the latter estimate has a wide confidence interval. Nevertheless, the SDDM produces good fits to both the survey and catch data (Figure E.2 in Appendix E). It seems that the combination of parameters used gives a good fit to the data, but the parameter estimates are highly uncertain. The similarities in stock biomass and fishing mortality trends and the overlap in the 95% credible intervals for B_0 and F_{msy} between the two models provide support in fitting the SDDM to the fish stocks in the Clyde.

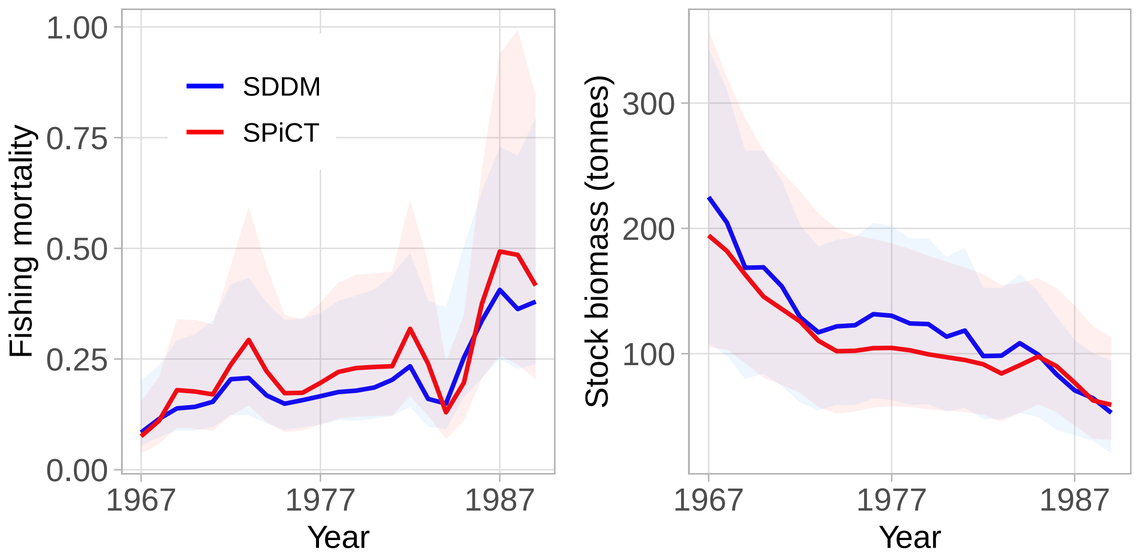

Figure 6.1: Fishing mortality (F harvest rate) and stock biomass (in tonnes) estimated for the SDDM and SPiCT models from the example albacore data. Shaded areas represent 0.95 credible intervals.

Table 6.4 - Median values and 95% confidence intervals of B_0 and F_{msy} estimated for the SDDM and mean estimates for the SPiCT model from the example albacore data.

Model	B_0	F_{msy}
SDDM	281.78 (127.87 - 212498.7)	0.16 (0.04 - 0.38)
SPiCT	201.48 (138.12 - 293.89)	0.37 (0.07 - 1.91)

6.3.2 Haddock data

Model fitting For the base model run, the model is able to estimate F_{msy} and B_0, although the distributions of the posterior estimates are not clearly defined (Figure 6.2 and 6.3). There seem to be two peaks in the joint distribution plot, which means that the model found more than one combination of parameters to explain the data. The model fits to the catch data very closely, with little variability given by the 95% credible intervals. The model produces a reasonable fit to the survey data, capturing the main trend of increasing abundance in mid-2000's (Figure 6.4). From Figure 6.5, fishing mortality fluctuates from one year to the next, but remains at high values between 1 and 1.5 for the whole time period. Stock biomass increases rapidly after 1990, reaching a peak between 2004-2006, and then decreases rapidly to low values for the last 10 years.

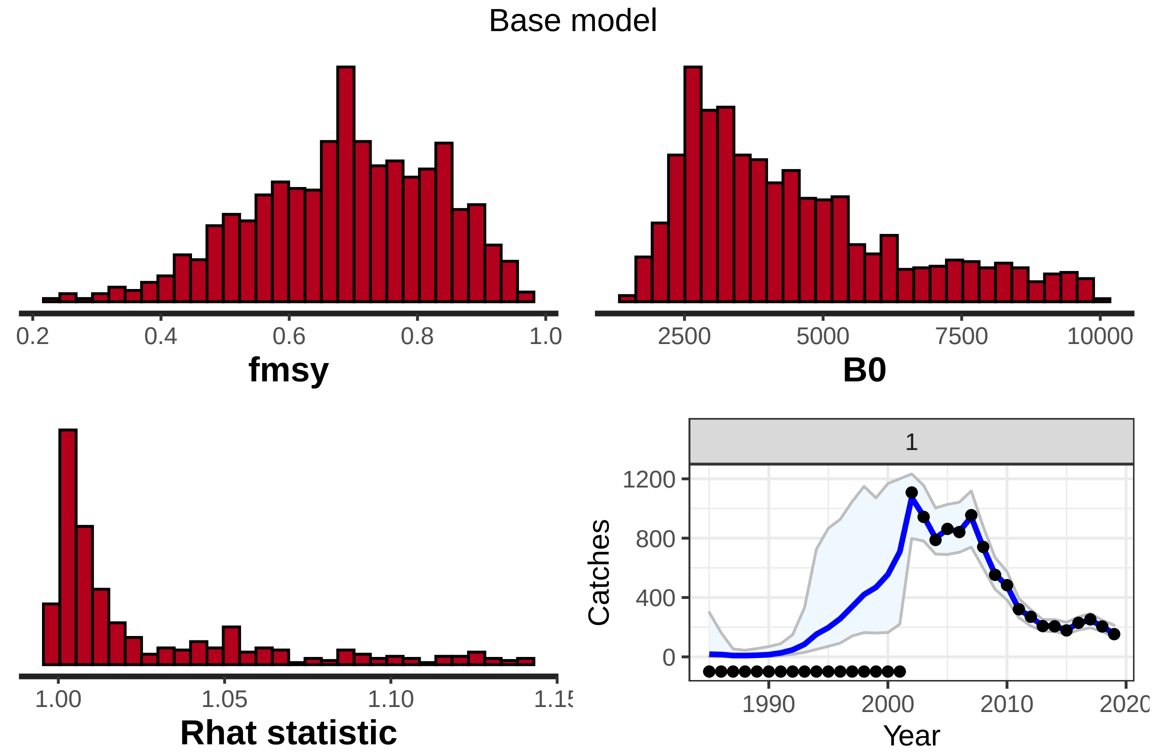

Figure 6.2: Histograms of Fmsy, Bo and Rhat value, and model fit to catch data for the base model applied to haddock data. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points. Data points before 2002 are missing values.

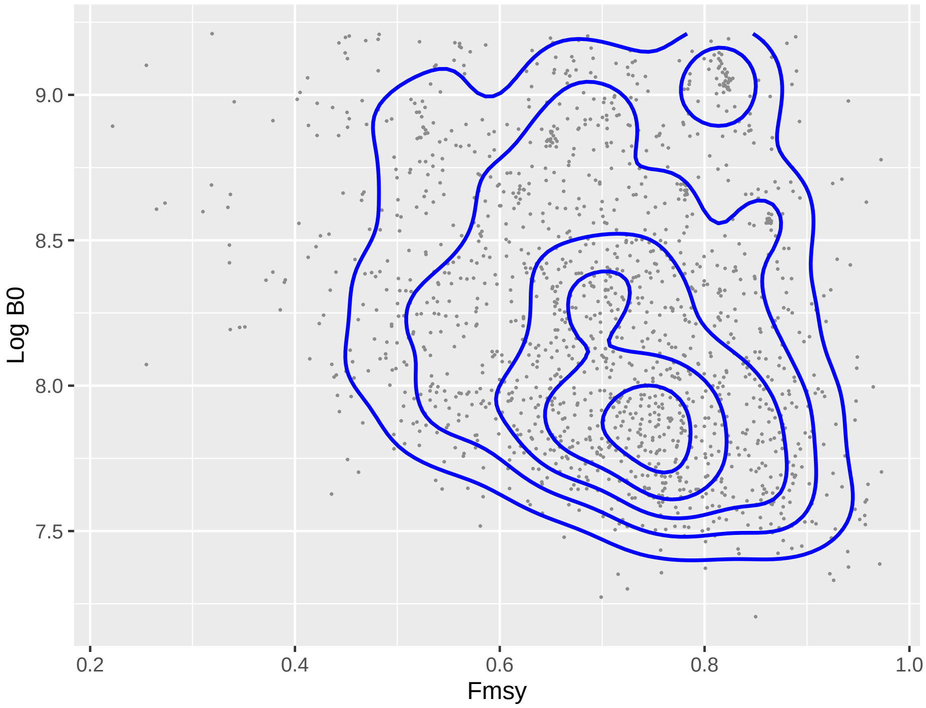

Figure 6.3: Joint distribution of Bo and Fmsy for the base model applied to haddock data. The contour lines represent probability density function.

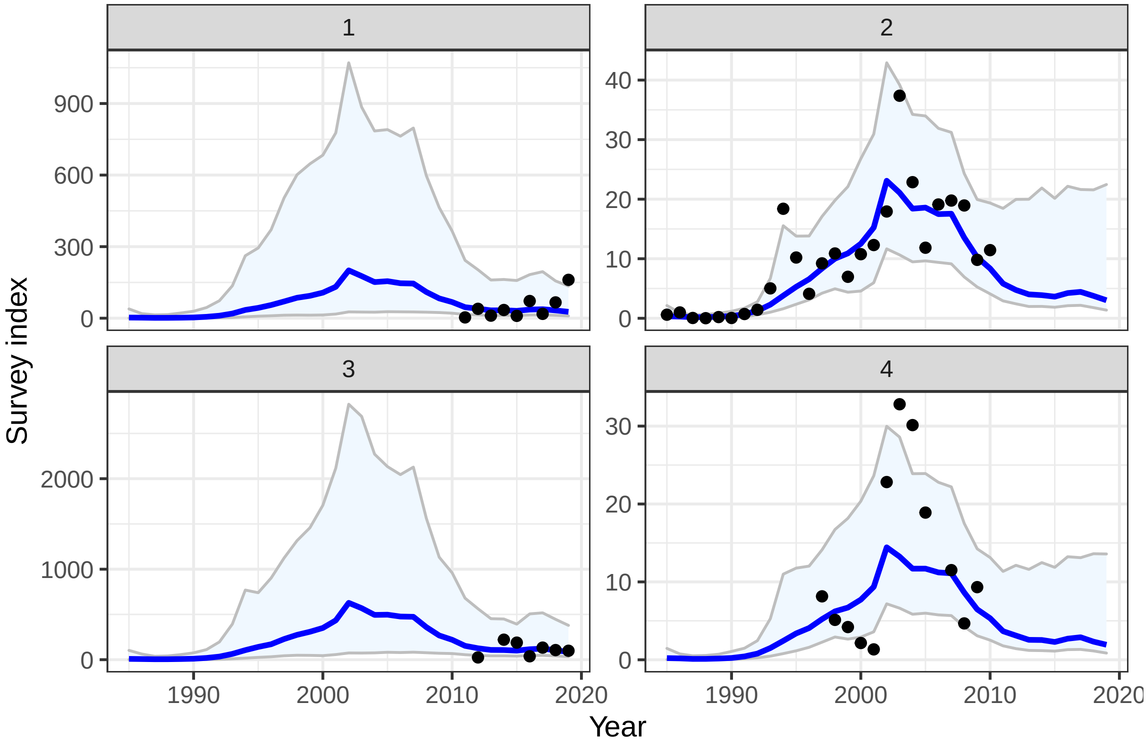

Figure 6.4: Model fit to survey data for the base model applied to haddock data. Each panel represents one survey time series. Values are in log scale. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points.

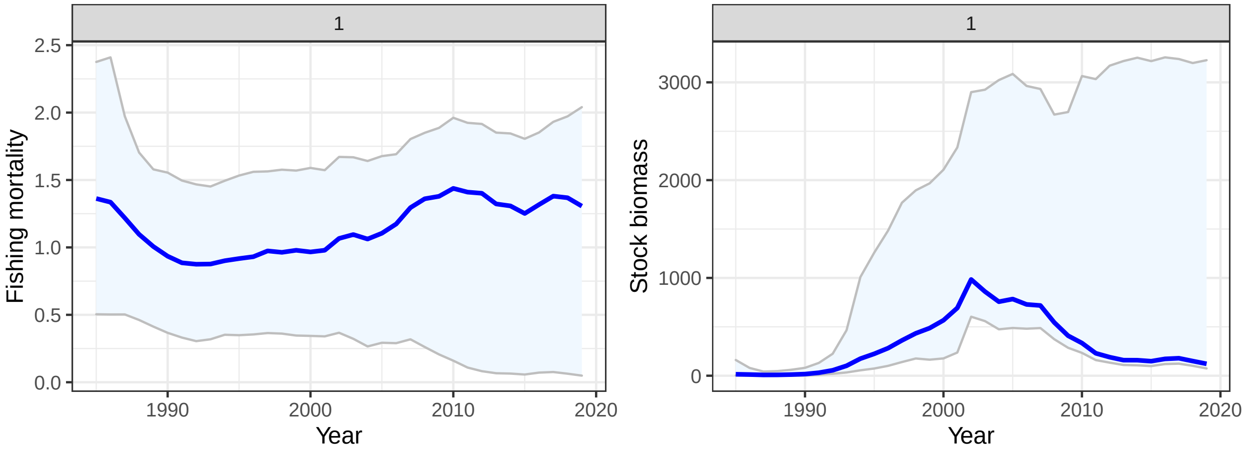

Figure 6.5: Estimated fishing mortality and stock biomass (in tonnes) for the base model applied to haddock data. Shaded area represents 0.95 credible interval.

Sensitivity analysis The fishing mortality and stock biomass trends are compared across all models runs in Figure 6.6. All models runs seem to show a decreasing trend of fishing mortality, except the base model and also run 1 to some extent. For run 1, the model estimates flat fishing mortality values from 1990 onward, with very wide 95% credible intervals (Figure E.3 in Appendix E). The base model run gives a slight decrease in F until 1990, but then steadily rises to peak values around 2010. Run 2 gives a substantial decrease in F until 1990, but after that it remains at values below 0.5 for the whole time series. For run 4, since the fishing mortality was calculated directly from the catches and biomass, the values are omitted before 2002.

In terms of stock biomass, each model run provides a different picture of biomass trend over the time period. For model run 1, the stock biomass was rescaled to the scale of the biomass for the base model so that the two time series start with the same biomass value. The same rescaling method was used for whiting and cod data. The biomass trend for run 1 is similar to the trend given by the base model, except for an increase of biomass at the end of the time series. Run 4 shows a peak in biomass in early 2000's, but then decreases to low values until 2010. After that, the biomass seems to increase slightly until 2019. Model run 2 gives an increase of biomass after 1990, after which it steadily increases and remains at around 5000 tonnes. The base model provides a completely distinct scale of biomass values, with very low values for the whole time series with a slight increase in biomass around early 2000's.

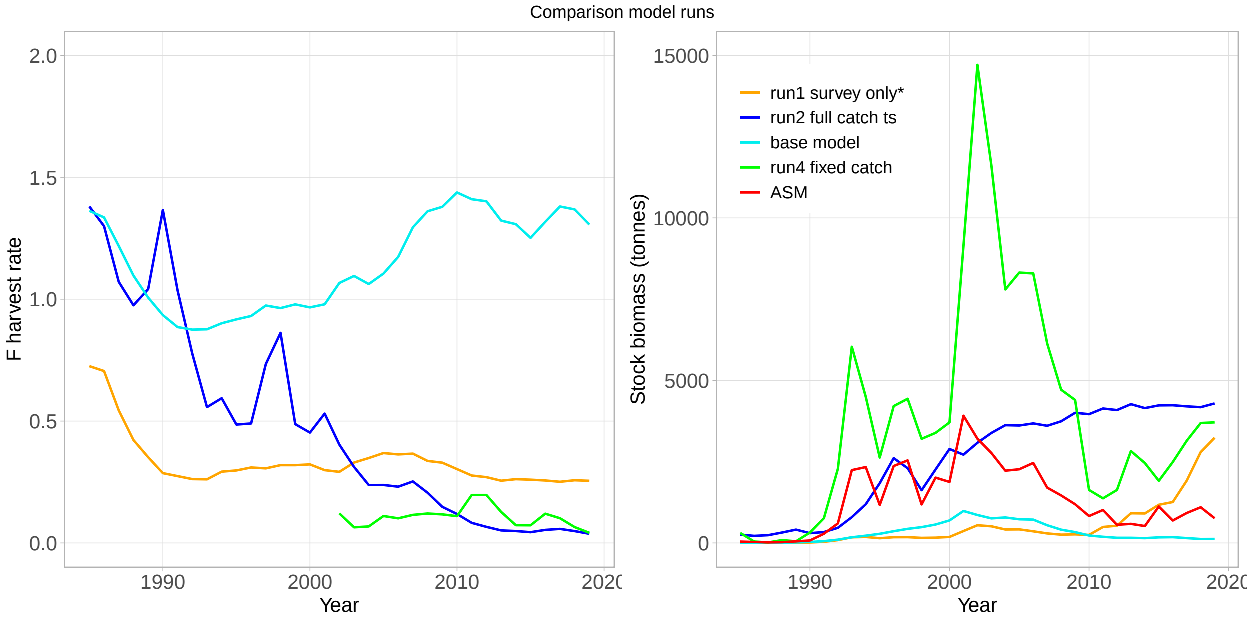

Figure 6.6: Fishing mortality and stock biomass estimated for each model run of the sensitivity analysis using haddock data. *rescaled values of stock biomass

The stock biomass from all model runs are also compared with the total stock biomass extracted from Chapter 5 for the age-structured model (ASM). The ASM stock biomass is plotted as the red line. Run 2 gives a completely different trend and scale of biomass than the ASM line. Run 4 provides a very similar pattern of biomass to the ASM line, but the scale is different, with run 4 estimating higher biomass values. The base model gives an overall smoother pattern of the biomass that is similar to the ASM line, but the biomass values are considerably lower.

When comparing the median values of the critical parameters, each model run provides a different set of values (Table 6.5). Model run 2 and the base model estimate values that are more closely related to each other, but overall very high values of F_{msy} (higher than 0.6). For run 4, F_{msy} is considerably lower than the other model runs.

Table 6.5 - Median values of B_0 and F_{msy} estimated for each model run using the haddock data.*B_0 value for run 1 has a different scale since biomass is at the scale of the survey index.

Haddock	B_0	F_{msy}
Run 1	2524.50*	0.29
Run 2	5459.76	0.66
Base model	3899.84	0.70
Run 4	7276.34	0.14

6.3.3 Whiting data

Model fitting For the base model run, the model is not able to properly estimate carrying capacity B_0, and the F_{msy} posterior distribution is left skewed and not well defined (Figure 6.7). The joint distribution shows more than one solution to the model (Figure 6.8). The model fits very closely to the catch data. The model fits to the survey data but it misses some main trends, such as the peak in abundance around 2000 (Figure 6.9). Fishing mortality is stable throughout the time series, with high values and wide 95% credible intervals (Figure 6.10). Stock biomass reaches a smooth peak around 2000 and then decreases to a minimum after that.

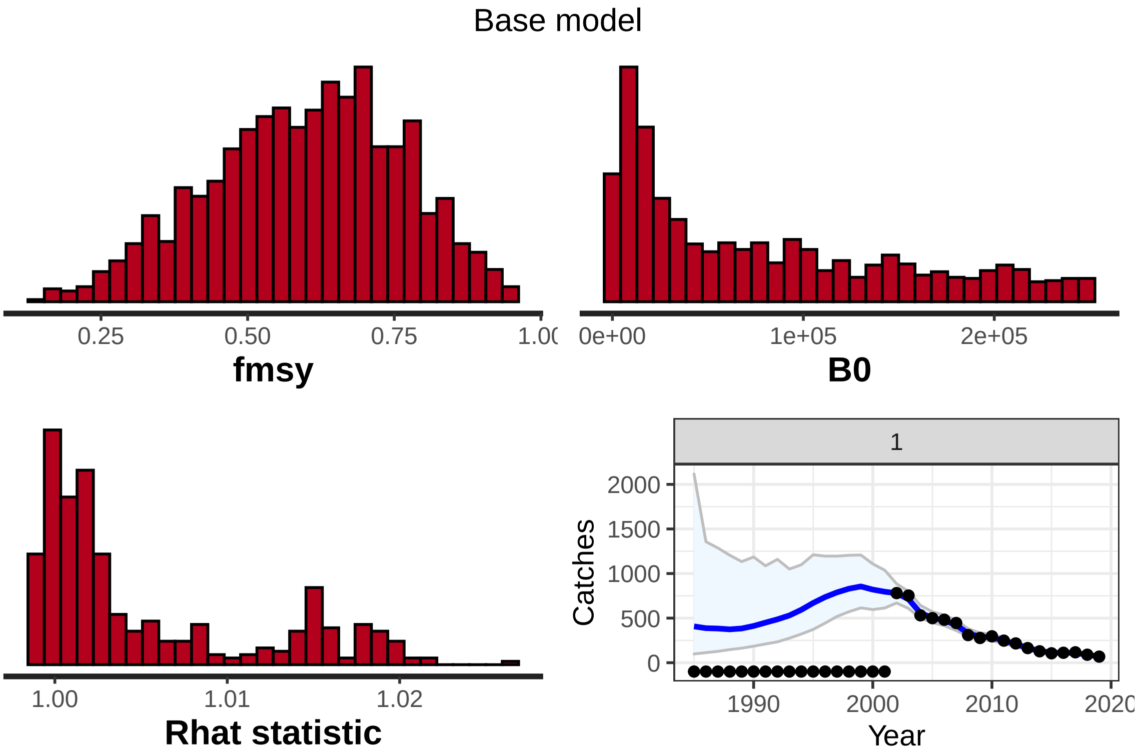

Figure 6.7: Histograms of Fmsy, Bo and Rhat value, and model fit to catch data for the base model applied to whiting data. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points. Data points before 2002 are missing values.

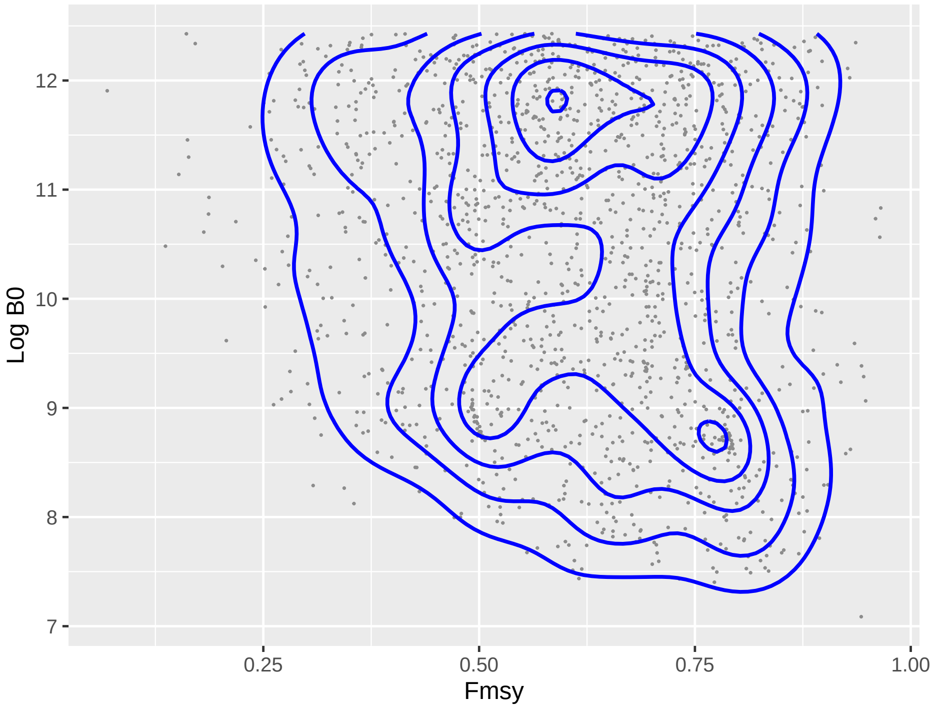

Figure 6.8: Joint distribution of Bo and Fmsy for the base model applied to whiting data. The contour lines represent probability density function.

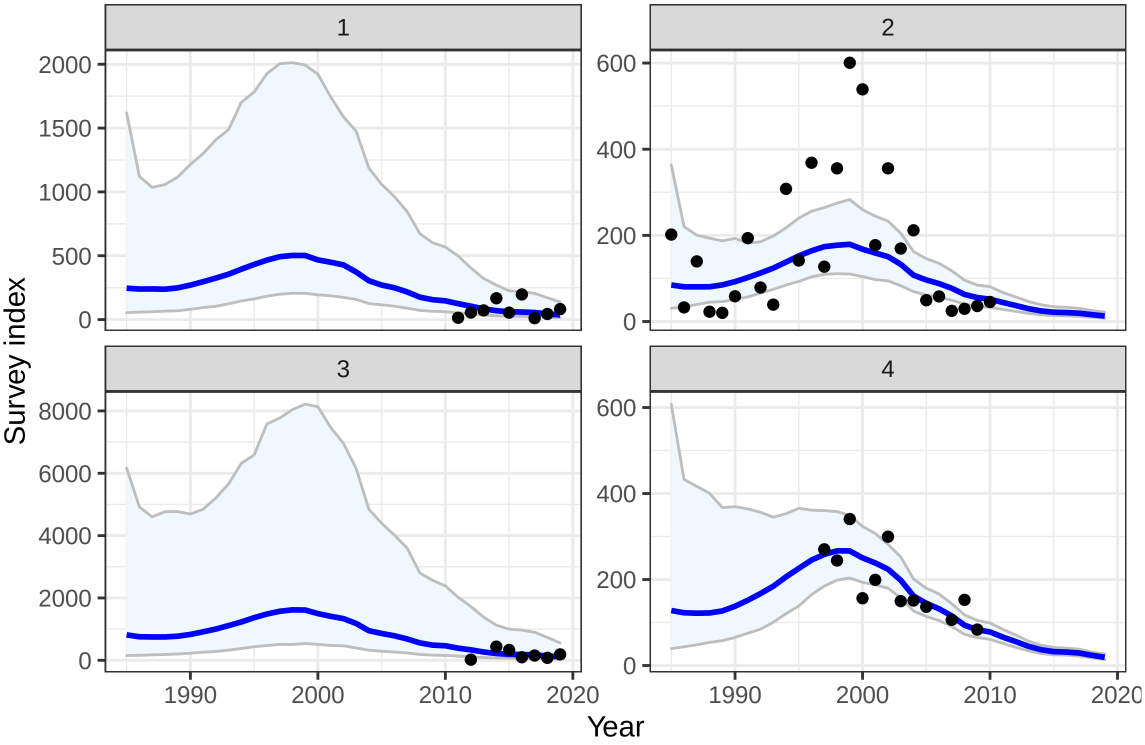

Figure 6.9: Model fit to survey data for the base model applied to whiting data. Each panel represents one survey time series. Values are in log scale. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points.

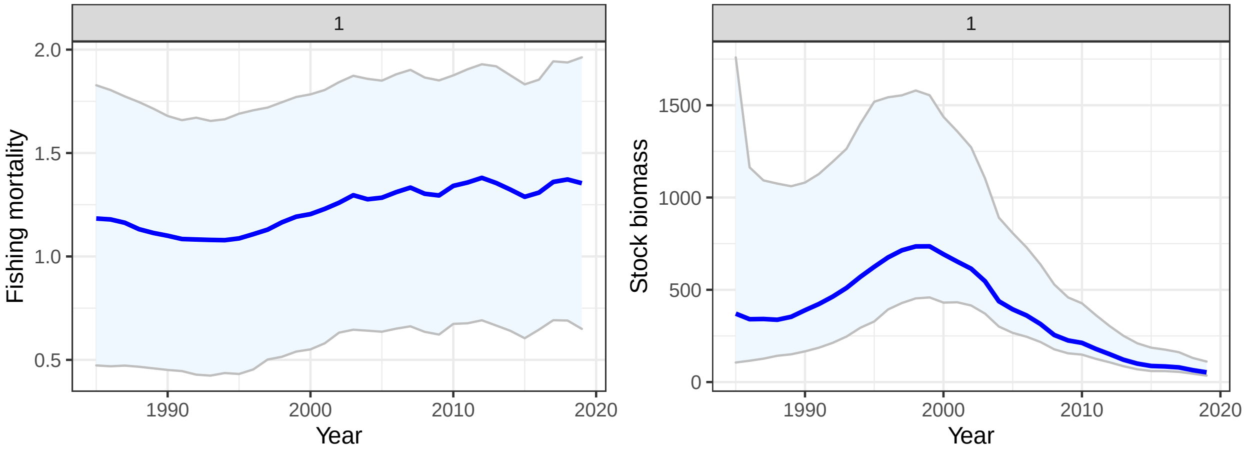

Figure 6.10: Estimated fishing mortality and stock biomass (in tonnes) for the base model applied to whiting data. Shaded area represents 0.95 credible interval.

Sensitivity analysis Fishing mortality and stock biomass trends for whiting are compared across all models runs in Figure 6.11. Most model runs show a relatively stable trend of fishing mortality throughout the time series, except for run 4, where the F values decrease after 2010. However, the scale of fishing mortality is very different between these model runs. Runs 2 and the base model give very high values of fishing mortality (F values between 1 and 1.5). In contrast, runs 1 and 4 give relatively low values of F (F lower than 0.5).

In terms of stock biomass, all model runs show a peak in biomass around 2000 followed by a decrease in biomass, although runs 1 and 4 show an increase in biomass for the most recent years. Run 1 provides a similar pattern of biomass as run 4, but the scale is very different between these two model runs.

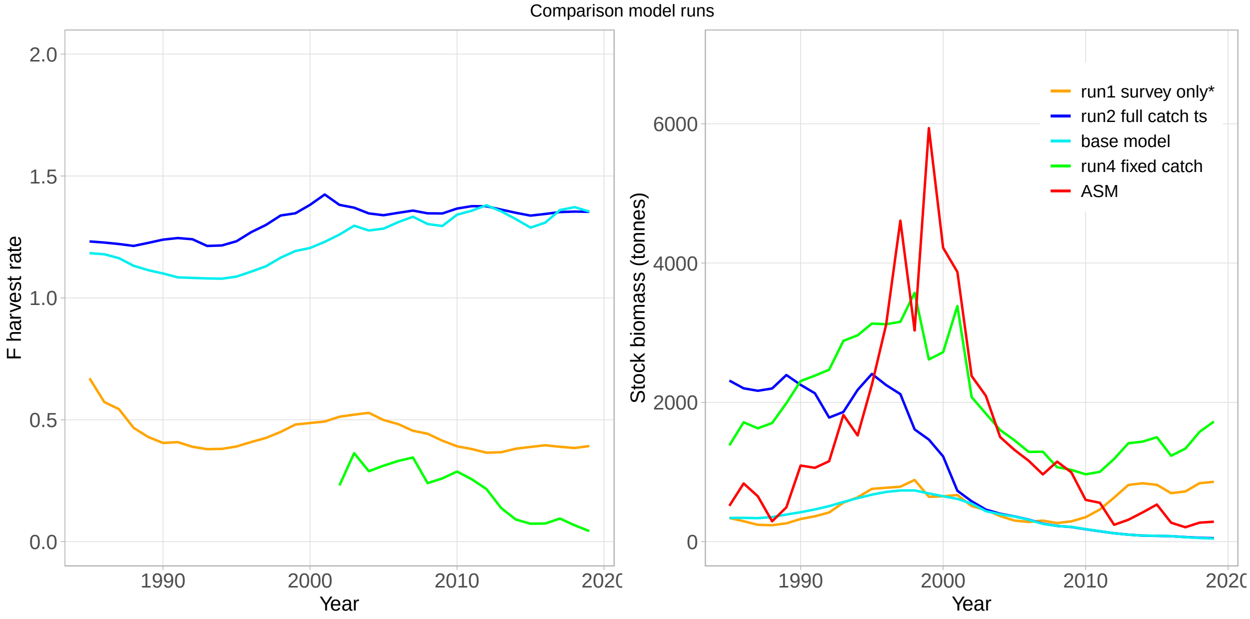

Figure 6.11: Fishing mortality and stock biomass estimated for each model run of the sensitivity analysis using whiting data. *rescaled values of stock biomass

The stock biomass from all model runs is compared with the total stock biomass extracted from Chapter 5 for the ASM (red line in Figure 6.11). The overall trend in biomass given by all runs coincides with the ASM line, particularly for run 4. The biomass trend starts low at the beginning of the time series, then increases to a maximum around 2000, and then decreases to minimum values for the most recent years. The scale of the biomass values overlaps between these model outputs, except for runs 1 and 4 that shows a slight increase in biomass after 2010.

Table 6.6 compares the median values of B_0 and F_{msy} between model runs. Model runs 2 and the base model estimate F_{msy} values that are more close to each other, with overall high values of F_{msy} (higher than 0.6). Runs 1 and 4 give lower values of F_{msy} than the other model runs.

Table 6.6 - Median values of B_0 and F_{msy} estimated for each model run using the whiting data.*B_0 value for run 1 has a different scale since biomass is at the scale of the survey index.

Whiting	B_0	F_{msy}
Run 1	2007.73*	0.40
Run 2	106377.9	0.62
Base model	55785.7	0.61
Run 4	2826.5	0.25

6.3.4 Cod data

Model fitting For the base model run, the model is not able to estimate B_0, and the F_{msy} posterior has a long left hand tail (Figure 6.12). The model has found that multiple combinations of B_0 and F_{msy} can satisfy the likelihood function, thus explaining the poorly defined marginal distributions for each parameter (Figure 6.13). The model seems to over fit the catch data, with almost non-existent 95% credible intervals from 2002 onward. The model gives a poor fit to the survey data (Figure 6.14). Fishing mortality stays relatively stable throughout the time series, at high values between 1 and 1.5 and wide 95% credible intervals (Figure 6.15). Stock biomass starts at high values in 1985, and then decreases rapidly, with some variability at low values after 2000.

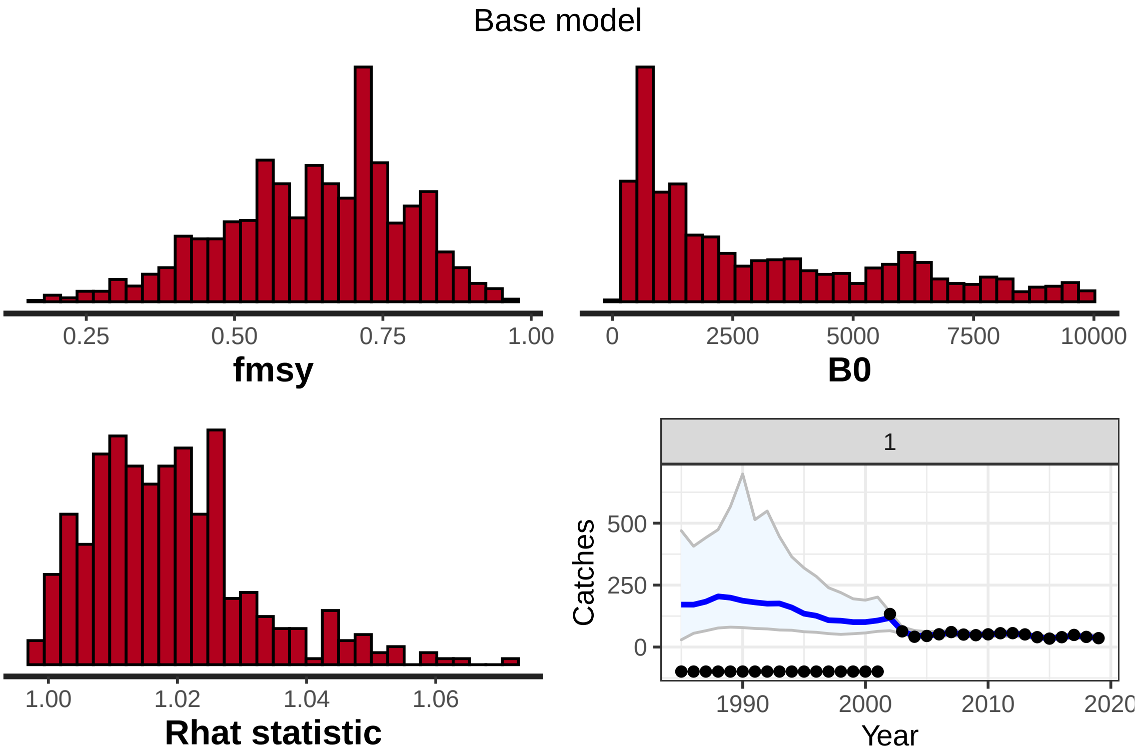

Figure 6.12: Histograms of Fmsy, Bo and Rhat value, and model fit to catch data for the base model applied to cod data. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points. Data points before 2002 are missing values.

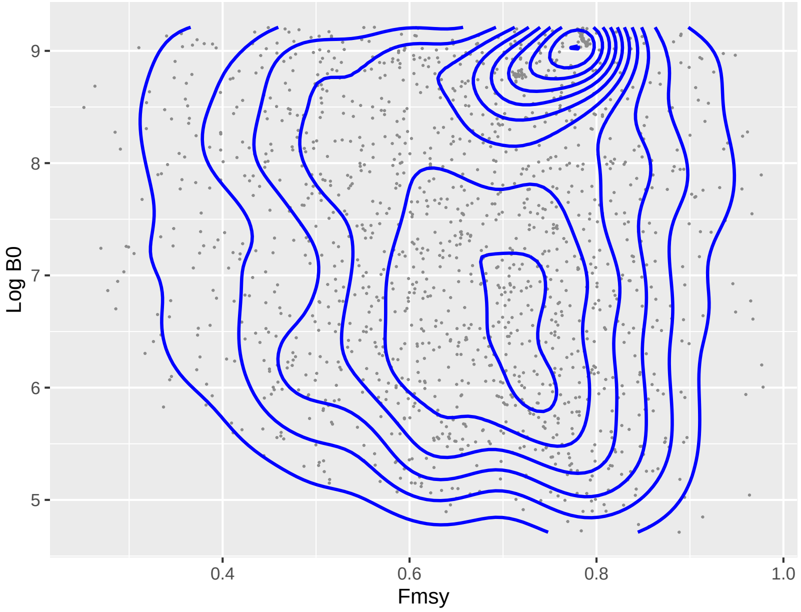

Figure 6.13: Joint distribution of Bo and Fmsy for the base model applied to cod data. The contour lines represent probability density function.

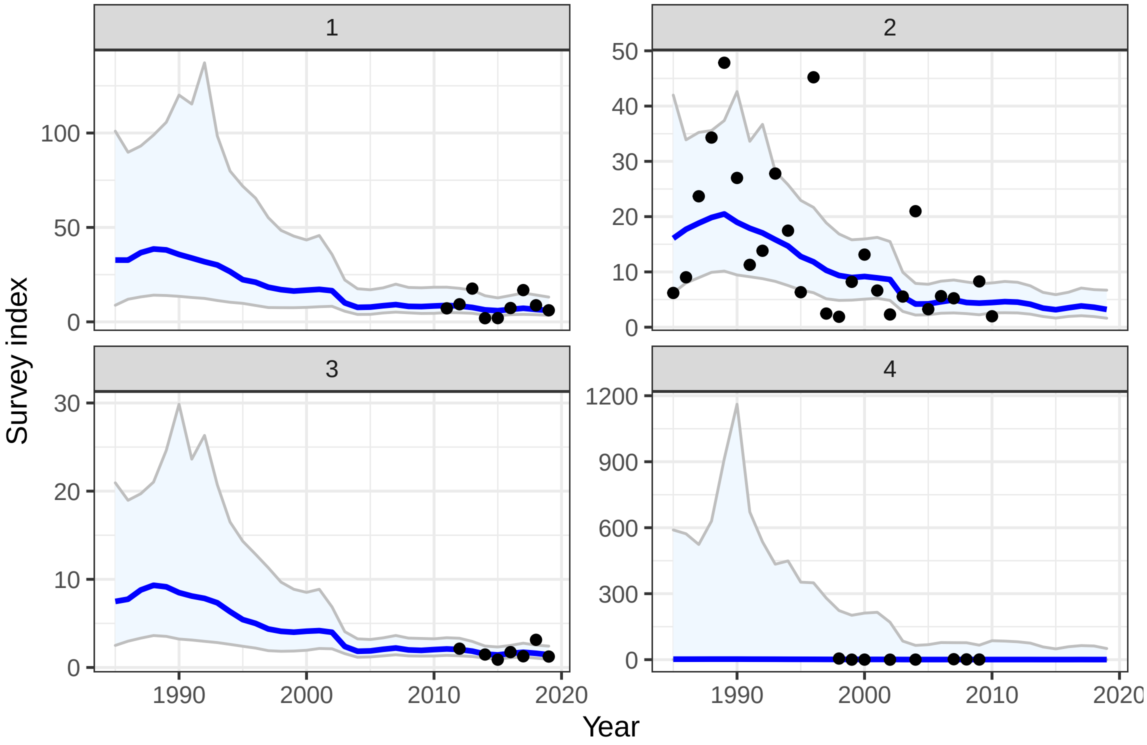

Figure 6.14: Model fit to survey data for the base model applied to cod data. Each panel represents one survey time series. Values are in log scale. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points.

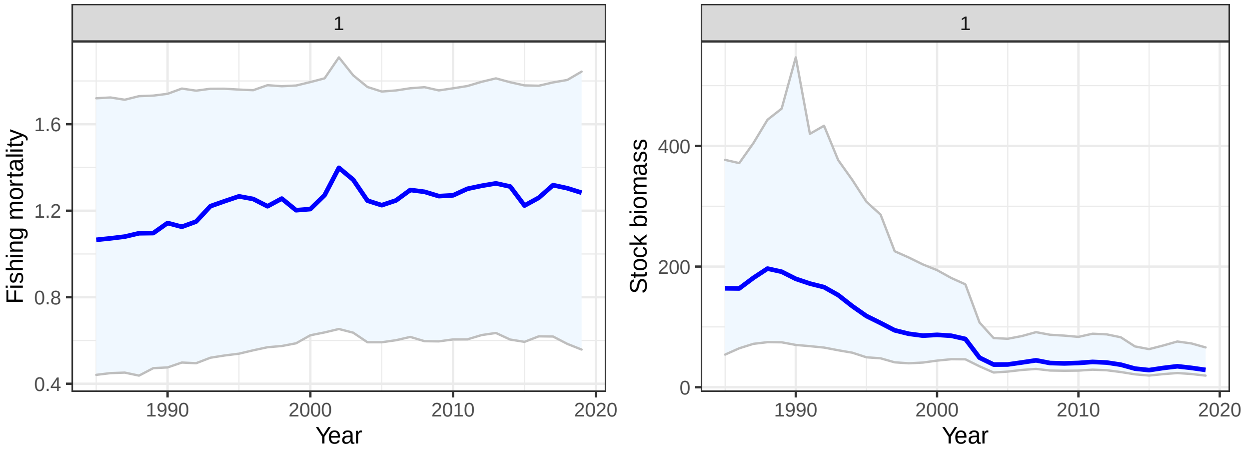

Figure 6.15: Estimated fishing mortality and stock biomass (in tonnes) for the base model applied to cod data. Shaded area represents 0.95 credible interval.

Sensitivity analysis Fishing mortality and stock biomass trends for cod are compared across all model runs in Figure 6.16. Most model runs show a relatively stable trend of high fishing mortality values throughout the time series. The scale of fishing mortality is different between model runs. Runs 2 and the base model give very high values of fishing mortality (F values between 1 and 1.5) compared with model runs 1 and 4 (F values below 0.5). Model runs 1 and 4 also give increases of stock biomass after 2015, in contrast with the other model runs.

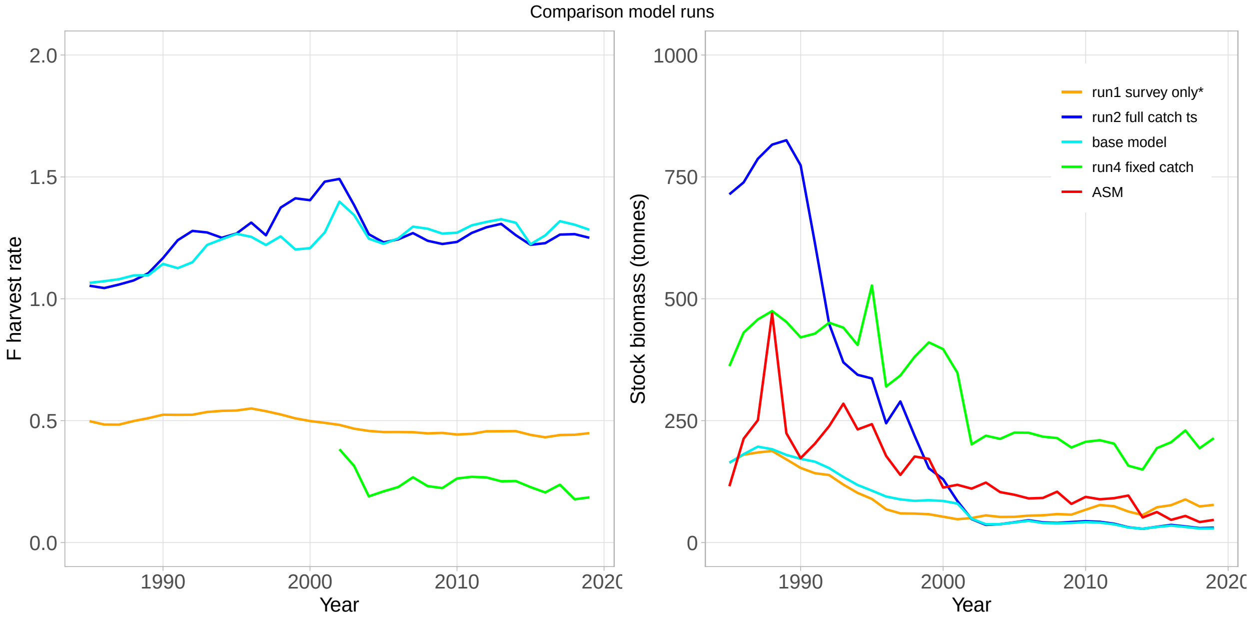

Figure 6.16: Fishing mortality and stock biomass estimated for each model run of the sensitivity analysis using cod data. *rescaled values of stock biomass

The stock biomass from all model runs is compared with the total stock biomass extracted from Chapter 5 for the ASM (red line in Figure 6.16). All model runs show a decreasing trend in biomass, that matches the trend from the ASM output. The ASM line lies between model run 4 and all the other model runs after 2000, and the biomass scale is very close to the SDDM outputs from 2010 onward (except for run 4).

Table 6.7 compares the median values of B_0 and F_{msy} between model runs. Each model gives a different value of carrying capacity B_0. Runs 1 and 4 give lower values of F_{msy} compared with runs 2 and the base model.

Table 6.7 - Median values of B_0 and F_{msy} estimated for each model run using the cod data.*B_0 value for run 1 has a different scale since biomass is at the scale of the survey index.

Cod	B_0	F_{msy}
Run 1	2272.71*	0.24
Run 2	6215.86	0.62
Base model	2051.73	0.66
Run 4	344.57	0.37

6.4 Discussion

A Bayesian Schaefer surplus production model (here called SDDM) developed for the Clyde stocks was applied to cod, haddock and whiting. Generally the SDDM does not provide satisfactory results. It does not fit well to the survey data and the critical parameters are not well estimated for most of the model runs. High uncertainty in parameter estimates is commonly found when fitting surplus production models (Hilborn and Walters, 1992).

For haddock, there was no model configuration that gave acceptable results. Only the base model (reduced catch) and run 4 (fixed catch) provided realistic outputs. However, the base model estimates stock biomass that is considerably lower than the biomass estimated by the ASM; and for run 4, the model estimates stock biomass at higher levels than the ASM outputs. As for the whiting results, none of the model runs provided an adequate fit to the survey data, not even run 1 which was fitted to the survey data only. This might be due to scaling issues between surveys. The scale of biomass from run 4 is very different than the other model runs. However, some of the model runs show a similar pattern and scale of biomass as the ASM results, which gives some degree of confidence of the overall biomass pattern and stock size. The results for cod have similarities to the whiting outputs. None of the model runs provided reasonable results. Only model runs 1-3 show a similar pattern and scale of fishing mortality over the time series. Nonetheless, all model runs estimate a decreasing trend of stock biomass, which has some similarity to the total biomass estimated by the ASM. Note, however, that the SDDM and the ASM use different sets of catch data.

The SDDM uses catches from the Nephrops and demersal fish fleet, while the ASM uses only the discards from the Nephrops fleet from 2002 onwards.

When comparing the results across species, the comparisons between model runs are more similar between whiting and cod, while for haddock each model configuration shows a different pattern. The models estimate lower values of carrying capacity for cod when compared with the other species. Given that run 2 and the base model seemed to be over fitting the catch data, the purpose of run 4 was to fix the catches as known in the model and explore if the model would try to give a better fit to the survey data. This seemed to work for the haddock data to some extent, with run 4 providing better model fits to the survey data, but not for whiting and cod. Also, this model (run 4) does not estimate fishing mortality. Instead, fishing mortality is calculated directly from the catches and biomass.

For most model runs, the critical parameters are not well estimated (skewed distributions). This means that the model cannot determine the scale of the stock biomass. In principle, fixing the catch data would have helped the model to pin down the scale of the biomass, but this is not the case for the results presented. It seems that the survey and the catches for most of the time series are not giving consistent patterns of abundance. As such, there is insufficient information in the data to distinguish between fishing mortality and biomass parameters. In the case of haddock, the assumptions in the SDDM are not consistent with the way the stock biomass increased in the Clyde. The SDDM assumes there is a stock-recruitment relationship, so one would expect lower recruitment at low stock size. It also assumes constant growth rate and carrying capacity over time. However, there was a substantial increase in recruitment of haddock in the Clyde around 1990's at low spawning stock sizes, likely from adjacent stocks. The implied stock dynamics are thus not consistent with the observations, so the SDDM does not work well for this stock. For cod and whiting, even though the model seems to perform better, the inconsistency between the survey and catch data prevents a final model configuration that provides reliable estimates of stock biomass and fishing mortality.

Although the SDDM fitted to the Clyde stocks did not produce satisfactory results, the model still estimates stock biomass trends that match to some extent the biomass patterns estimated from the ASM in Chapter 5. The biomass trends for whiting and cod decreased mainly from 2000 onward to minimum values. For haddock there seems to be also a decrease in biomass after 2000, but with a recent increase between 2015 and 2019.

In the following Chapter, the Clyde stocks are projected into the future and the ASM outputs were chosen for this analysis. This is because the ASM makes no strong assumptions about the stock-recruitment relationship and is more robust to changes in alternative model configurations. The ASM also makes use of informative age data. The SDDM is very sensitive to the input catch data and is not able to estimate critical parameters adequately, so it would not be a good option for drawing future scenarios of fishing mortality and biomass trends.

CHAPTER 7 - Scenario Analysis

7.1 Introduction

Previous chapters show that demersal fish stocks in the Clyde have been declining for the past 20 years, and this appears to be the result of high exploitation rates. In this chapter, scenarios of reduced fishing mortality were explored to see how this influences potential stock recovery. For doing so, a model that projects the population into the future is needed. The ASM was used for this analysis, but only describes how cohorts decline over time and does not predict future recruitment. Modelling the stock-recruitment relationship is therefore an essential element of the projection model to be considered in this chapter.

For many stocks, there is a relationship between the size of the spawning stock (mature fish biomass) and the number of young produced by that spawning stock (Hilborn and Walters, 1992). Understanding this relationship is important to ensure that the stock won't crash due to recruitment failure (also known as recruitment overfishing). Stock-recruitment models assume that at low stock sizes, recruitment is primarily driven by density-independent factors and increases monotonically with stock size (Subbey et al., 2014). However, at large stock sizes, other factors (density-dependent effects) are more influential on the survivorship of the young fish, and either recruitment decreases at high stock size (Ricker curve) or reaches an asymptote (Beverton-Holt curve; Hilborn and Walters, 1992). In fact, recruitment is linked not only to the amount of spawning biomass, but also to other factors such as parental size, growth history and environmental conditions of each individual fish (Subbey et al., 2014).

Reference points are one of the main pillars of current fisheries management advice and are often conditioned by the stock-recruitment relationship. Biological and economic reference points are used to compare the current status of a stock or fishery against a desirable (or undesirable) state; they provide guidance for management in both evaluating the current status of a stock and the possibilities of exploring it.

There are three main types of reference points: limit reference points, target reference points and trigger/threshold reference points (Lart, 2022; ICES, 2017b; ICES, 2022c). Limit reference points define an undesirable biological stock status. To keep the stock in safe conditions, the probability of crossing a limit reference point should be low. Limit reference points are often based on the biology of the stock and can be set to avoid recruitment overfishing. A target reference point defines the ideal fishery state that should be achieved and maintained in the long term with a high degree of certainty. A target reference point can be based on other aspects besides the biology of the stock, such as social, ecological, or economic considerations. A trigger reference point is usually set between the target and limit reference points, and triggers a pre-defined management response to help the fishery remain close to the target reference point and avoid breaching the limit reference point.

Within the ICES framework, the precautionary approach sets reference points based on stock biomass. The most commonly used is a limit reference point on spawning biomass (B_{lim}), below which a stock is considered to have impaired reproductive capacity. On the other hand, the maximum sustainable yield (MSY) approach is directed to reference points related with fishing mortality. The MSY concept is based on harvesting the stock at optimal levels to maximize yields in the long-term. The most used reference point is fishing mortality that leads to MSY (F_{msy}), and a corresponding stock biomass (B_{msy}). Nowadays, F_{msy} is set as the limit level of fishing mortality that the stock should be exposed, while B_{msy} can be used as a target reference point (ICES, 2022c).

Reference points can be calculated directly from model parameters as is the case of surplus production models, shown in Chapter 6. However, in more complex age-based assessment models, reference points are usually estimated by a post hoc analysis that involves visual inspection of the stock-recruitment relationship to determine B_{lim}, and a long-term forecast simulation for estimating F_{msy} (Albertsen and Trijoulet, 2020; ICES, 2022e). As a consequence, values of the reference points are highly dependent on the biological quantities used in the age-based assessment models (Kraak et al., 2005). Additionally, most stock-recruitment relationships have the assumption of stationarity and so do not effectively deal with shifts in productivity of the stock (Subbey et al., 2014; Zhang et al., 2021). In the ICES system, a benchmark process occurs periodically (i.e. every 3 to 5 years), which consists of reviewing the data, the assessment model and the reference points for each stock, thus updating the management advice to the current situation (ICES, 2022c).

In this chapter, the analysis was performed in two stages. First, using the results of the age-structured model from Chapter 5, we extracted essential population parameters such as current stock size (spawning stock biomass) and fishing mortality. Second, we used these values to calculate conventional equilibrium reference points such as F_{msy} and B_{msy} and projected the populations into the next 30 years under differing assumptions of fishing mortality and recruitment scenarios. The projections were done using a model that takes account of uncertainty in the model parameters, recruitment stochasticity, and the structural relationship between stock size and recruitment (annual number of fish at age 0).

7.2 Methods

7.2.1 Projection model

For the projection model, the population equations from the ASM described in Chapter 4 were used and the parameters and outputs from Chapter 5 for the reference model were also used. The populations were projected into the future according to different fishing scenarios and recruitment regimes, separately for cod, haddock and whiting. A random subset of 1000 samples was previously saved from the Markov Chain Monte Carlo (MCMC) simulations for all model parameters from the ASM for subsequent use in the projection model described below.

Considering 2019 as the starting year, equations 4.3-4.5 (Chapter 4) were used to project the population forward 30 years by which time the populations have reached an equilibrium. The input quantities used for projecting the population are listed in Table 7.1. A plus group of fish at age 5 and older was added within the projection code to account for fish that would survive at older ages, that are relevant for scenarios of reduced fishing mortality. Mean weights at age and natural mortality values from 2019 were used.

Table 7.1 - Input quantities for the simulation model extracted from MCMC samples after fitting the ASM.

Quantity	Description	Usage
mat_a	Proportion of fish mature at age class	Calculation of SSB
wt_a	Mean weight at age of an individual fish	Calculation of SSB
M_a	Natural mortality	Calculation of total mortality
$N_{a,2019}$	Population across ages in 2019	Basic population for projections
sel_a	Fishery selectivity at age	Calculation of fishing mortality
$f_{y,2019}$	Fishing effort multiplier in 2019	Reference fishing effort for projections
σ_f^2	Process error on fishing effort	Random variability added to fishing effort
SSB_y	Spawning stock biomass in all years	Estimation of stock-recruitment parameters
$N_{0,y}$	Recruitment at age 0 in all years	Estimation of stock-recruitment parameters

Each realization of the projections was based on one of the MCMC samples. For each sample, a Beverton-Holt stock-recruitment model was fitted by least squares to the full time series of stock-recruitment pairs (equation 7.1) with lognormally distributed errors. A common re-parameterized formulation (Haddon, 2021) was used such as:

$$R = \frac{aSSB}{b + SSB} e^{\varepsilon_t} \tag{7.1}$$

with

$$\varepsilon_t \sim Normal(0, \sigma_R^2) \tag{7.2}$$

where a is maximum recruitment and b is the spawning stock needed to produce, on average, half maximum recruitment ($a/2$).

A Ricker stock-recruitment model was also tested which took the form:

$$R = \alpha SSBe^{-\beta SSB}e^{\varepsilon_t} \tag{7.3}$$

with α representing the recruits-per-spawner at low stock levels, β relates to the rate of decrease of recruits-per-spawner as SSB increases, and ε_t indicates log-normal residual errors. Annual recruitment variability was simulated by adding one random lognormal residual to each of the fitted recruitment values.

Using a Ricker stock-recruitment model gave identical results to the Beverton-Holt (BH) model since the estimated spawning biomass levels are relatively low. Therefore the results are shown only using a BH curve.

For investigating how changes in fishing mortality affect stock productivity, the population was projected under 6 different levels of fishing and 4 recruitment scenarios. The current level of fishing mortality estimated for 2019 was considered as a baseline projection, and recruitment from the BH stock-recruitment model for the full time series of stock-recruitment pairs (1985 - 2019). For fishing mortality, the 2019 estimate was scaled by decreases of 25, 50, 75% and 99% (no fishing activity) and an increase of 25%. For the 4 recruitment scenarios, the recruitment from the baseline projection was inflated by 50 and 100% (doubling the recruitment level) or deflated by 50% to represent variability in environmental conditions that would favor recruitment or inhibit large year classes. Summary statistics were calculated such as median SSB, recruitment, catches and corresponding credible intervals under the various scenarios.

7.2.2 Reference points - equilibrium analysis

Maximum sustainable yield (MSY) and associated reference points were calculated for each MCMC sample in a deterministic way using the stock-recruitment curve and “per-recruit” analysis. For a constant fishing mortality rate, the population will stabilize at an equilibrium biomass, if the annual number of recruits and subsequent survival are enough to sustain the population. This is illustrated in Figure F.1 in Appendix F. A stock-recruitment curve and a replacement line (with slope equal to the inverse of SSB/R) intersect at an equilibrium point to which the population is attracted to. The equilibrium spawning biomass (SSB_{eq}) can be calculated as:

$$SSB_{eq} = a\ SPR - b \tag{7.4}$$

where a and b are parameters estimated from the BH model. SPR is the spawning biomass per recruit, and using the mean weight at age (wt_a), maturity at age (mat_a) and a cumulative total mortality per recruit/fish as it progresses along the cohort (Z_i) can be calculated as:

$$SPR = \sum_{a=0}^{A} wt_a mat_a e^{-\sum_{i=0}^{a-1} Z_i} \tag{7.5}$$

The equilibrium recruitment (R_{eq} in numbers of fish) is calculated as:

$$R_{eq} = \frac{SSB_{eq}}{SPR} \tag{7.6}$$

and equilibrium yield (Y_{eq}) as

$$Y_{eq} = R_{eq} YPR \tag{7.7}$$

and YPR is the yield-per-recruit (from the Baranov catch equation):

$$YPR = \sum_{a=0}^{A} \frac{W_a F_a e^{-\sum_{i=0}^{a-1} Z_i}(1 - e^{-Z_a})}{Z_a} \tag{7.8}$$

For each draw of the fitted parameters, an optimization routine to find the fishing effort multiplier that maximizes yield (Y_{eq}) was applied. This resulted in distributions of MSY (maximum sustainable yield), F_{msy} (fishing mortality at MSY), B_{msy} (spawning stock biomass at MSY) and Rmsy (recruitment at MSY) from all MCMC samples. The median and 95% credible intervals were computed for each reference point.

7.2.2 Reference points - simulation method

Another way to calculate MSY is by using simulations, where the stock is projected forward until it reaches equilibrium over a range of fishing mortalities, and then finding the fishing mortality value that gives the highest mean catch at equilibrium. To implement the simulation method, a set of multipliers was chosen for each species to scale the fishing mortality values (fishing mortality was scaled by values ranging from 0.1 to 2, in increments of 0.1 for haddock and whiting, and increments of 0.05 for cod). For each of the MCMC samples, the projection was run to equilibrium (30 years) for a given F multiplier. The median catch and biomass were calculated for the last 10 years of the projection and stored, and this process was done for each of the F multipliers. Plotting the median catch against the F multipliers will provide the equilibrium yield curve, and the maximum of this curve corresponds to MSY (Figures F.1, F.3 and F.5 in Appendix F). A lowess smoother was used to estimate the maximum value of the yield curve and to obtain the F multiplier corresponding to MSY. With this information, the B_{msy} and the other reference points could also be extracted.

The B_{msy} from the MSY projection was compared with the B_{msy} calculated from the equilibrium analysis, and the distributions of B_{msy} were plotted after applying a kernel density smoother (Figures F.2, F.4 and F.6 available in Appendix F). Since only the last 10 years of the projections were used, the distributions of B_{msy} are representative of the population at median values from 2038 to 2048 (after reaching equilibrium).

To investigate the conditions for recovery of the populations, the probability of recovery increasing above the B_{msy} level at year 5 and year 30 of the projections was calculated. The B_{msy} estimated from the MSY projection was used here since it is estimated within the population projections. This probability was defined for each simulation (n) as:

$$Prob.recovery(n) = \begin{cases} 1 \ for \ SSB > B_{msy} \\ 0 \ for \ SSB \leq B_{msy} \end{cases} \tag{7.9}$$

and then probability of recovery at year 5 (or year 30) calculated by summing across simulations and dividing by the total number of simulations:

$$Prob.recovery\ year\ 5\ (or\ year\ 30) = \frac{\sum Prob.recovery(n)}{nsimul} \tag{7.10}$$

7.3 Results

7.3.1 Haddock projections

Table 7.2 shows the estimated reference points from the equilibrium analysis. The first 5 years of data had to be excluded from the stock-recruitment curve since there was almost no spawning biomass of haddock in the Clyde until 1990. Biomass at MSY (B_{msy}) was estimated at around 207 tonnes, and fishing mortality corresponding to F_{msy} (F_{msy}) was 0.63.

Table 7.2 - Estimated references points (median and quantiles) from equilibrium analysis for haddock data.

Quantile	B_{msy}	F_{msy}	MSY	Rmsy
2.5%	77.97	0.43	85.91	10026.31
50%	207.33	0.63	128.19	15726.45
97.5%	525.15	0.96	219.61	28798.71

Table 7.3 summarizes the reference points extracted from the simulation method. The equilibrium biomass is slightly lower than the one calculated previously in Table 7.2. This might be related with the fact that the F_{msy} value estimated from the simulated method is also slightly lower.

Table 7.3 - Estimated references points (median and quantiles) from simulation method for haddock data.

Quantile	Bmsy	Fmsy	MSY	Rmsy
2.5%	21.02	0.13	32.11	1465.87
50%	182.36	0.50	127.73	15007.17
97.5%	891.66	1.26	425.60	78363.88

The population was projected for 30 years using the baseline year of 2019, with one of the scenarios assuming fishing mortality would remain at the same level. The outputs are shown in Figure 7.1, together with the estimates from the ASM between 1985 and 2019. Assuming a stock-recruitment relationship that follows a Beverton-Holt curve, the population is projected to stabilize at values of 200 tonnes stock biomass and 15 400 recruits per year. The projected stock-recruits pairs are represented by the red line in the stock-recruitment plot. The value lying outside the curve corresponds to the stock-recruit pair of 2019. The population is projected at stable recruitment values from 2022 onward, with wide credible intervals. The mean F plot shows the value of mean fishing mortality over ages 1 to 3. After 2019, the population was projected at a constant value of fishing mortality, but with stationary process noise. The current level of F is below the F_{msy} reference point. The last panel plot shows the stock biomass trend across the whole time series, and after 2025 the projected population stabilizes at a value of biomass around 200 tonnes (grey line). This level of biomass is approximately similar to the B_{msy} levels.

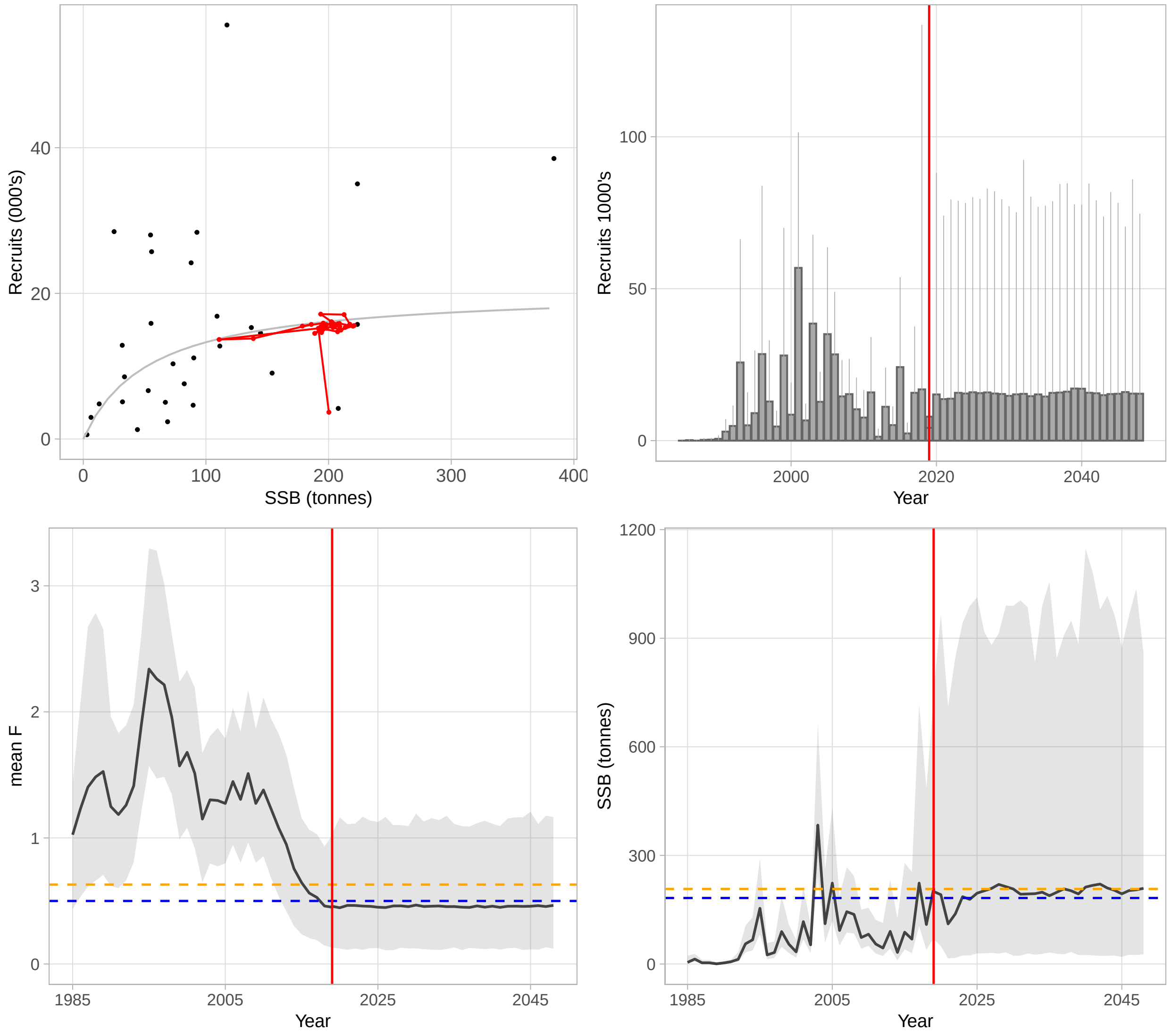

Figure 7.1: Outputs of the projected haddock population at current fishing mortality levels, combined with population estimates from 1985 to 2019. Top left plot: stock-recruitment relationship, black dots representing estimates between 1985-2019 and associated Beverton-Holt curve, and projected median S-R as the red dotted line. Top right plot: annual recruitment with errors bars representing 0.95 credible intervals, projection starts in 2019. Bottom left plot: mean fishing mortality, shaded area represents 0.95 credible intervals; red vertical line marks the start of the projection. Bottom right plot: spawning stock biomass (SSB), projected biomass starts in 2019. Orange dashed lines represent reference points estimated from equilibrium analysis and blue dashed lines represent reference points from the simulation method.

Probability of recovery and stock biomass are shown as a function of relative fishing mortality for the different recruitment scenarios (Figure 7.2). The probability of recovery is defined as the probability of the biomass in a certain year being above B_{msy} levels (using the B_{msy} estimated from the simulation method). The relative F is the fishing effort multiplier used for the fishing scenarios at different levels. When relative F is 1, it corresponds to the current level of fishing mortality (fishing mortality for 2019). When relative F is 1.5, the 2019 fishing mortality value was scaled by an increase of 50%. As expected, the probability of recovery decreases as relative F increases. For a relative F of 0, the probability of recovery above B_{msy} levels is 1, and vice-versa (when relative F is 1.5 there is less than 50% probability of recovery). At current levels of F (relative F of 1), the probability of recovery is 0.5 in the short-term projection (year 5) and just above 0.5 in the long-term projection (year 30). The current levels of fishing mortality are below F_{msy}. In terms of spawning stock biomass, at current levels of F, SSB is close to 200 tonnes which is approximately the same level of B_{msy}. There are subtle differences between year 5 and year 30 of the projections, with slightly higher biomass values at year 30, particularly for scenarios with increased recruitment levels (green lines in plot). In a scenario of doubling the recruitment levels, the probability of recovery would increase to almost 75% in 5 years and around 85% in 30 years. In contrast, a scenario of poor recruitment (reduced by half) would mean a decrease in the probability of recovery to less than 25%.

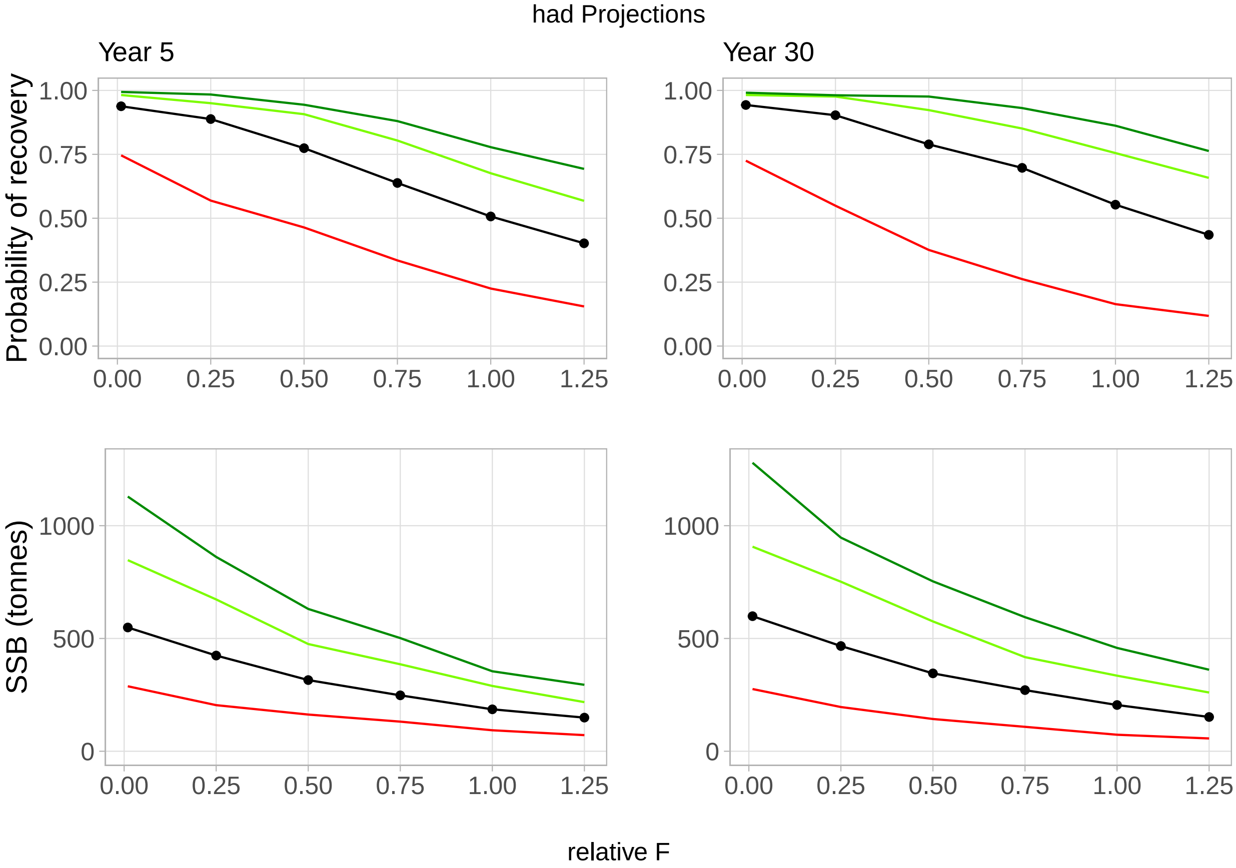

Figure 7.2: Comparison of short-term and long-term projections on the probability of stock recovery and median spawning stock biomass (SSB) for haddock data. Projections were performed at varing levels of fishing mortality and recruitment scenarios. The black dotted line represents baseline recruitment; the red line represents a 0.5 decrease in recruitment; the light green and dark green lines a 1.5 and 2x increase in recruitment, respectively.

7.3.2 Whiting projections

The estimated reference points from the equilibrium analysis are shown in Table 7.4. Biomass at MSY (B_{msy}) was estimated at almost 200 tonnes and fishing mortality corresponding to F_{msy} at a high value of around 0.89.

Table 7.4 - Estimated references points (median and quantiles) from equilibrium analysis for whiting data.

Quantile	B_{msy}	F_{msy}	MSY	Rmsy
2.5%	66.55	0.70	148.36	17340.44
50%	198.94	0.89	312.92	38318.37
97.5%	627.54	1.22	744.00	96714.65

Table 7.5 summarizes the reference points extracted from the simulation method. The B_{msy} is higher than previously shown in Table 7.4. This might be explained by the change in the peak of the distribution of the B_{msy} values and a more skewed distribution of B_{msy}, shown in Figure F.5 (Appendix F). The value of F_{msy} is slightly lower than in Table 7.4.

Table 7.5 - Estimated references points (median and quantiles) from simulation method for whiting data.

Quantile	Bmsy	Fmsy	MSY	Rmsy
2.5%	32.51	0.37	90.53	3612.59
50%	248.41	0.80	377.31	41424.79
97.5%	1441.85	1.34	1517.40	257383.17

The population was projected for 30 years using the baseline year of 2019. The outputs are shown in Figure 7.3, together with the estimates from the ASM between 1985 and 2019. Assuming a stock-recruitment relationship that follows the BH curve, the population is projected to stabilize at values of around 40 tonnes of SSB and recruitment of 19 000 fish per year. The projected stock-recruit pairs are represented by the red line in the stock-recruitment plot. The left hand side of the red line corresponds to the start of the projected trajectory. The population is projected at stable recruitment values from 2022 onward, with wide credible intervals. In the mean F plot, the population was projected at a constant current value of fishing mortality after 2019 (including process noise). The current level of F is considerably higher than the F_{msy} reference points. The SSB plot shows that biomass under a projection of maintaining F levels is much lower than the B_{msy} reference points.

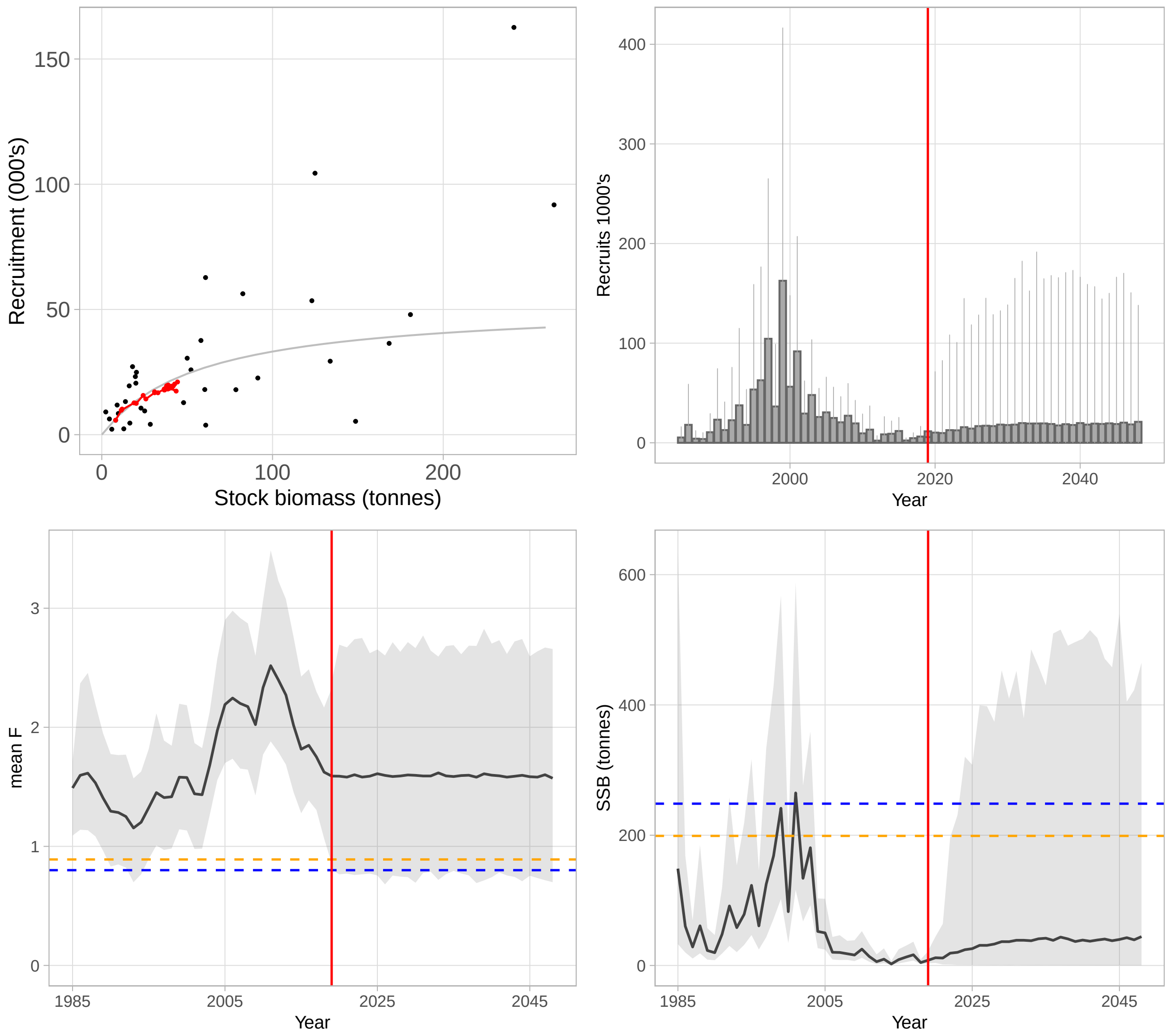

Figure 7.3: Outputs of the projected whiting population at current fishing mortality levels, combined with population estimates from 1985 to 2019. Top left plot: stock-recruitment relationship, black dots representing estimates between 1985-2019 and associated Beverton-Holt curve, and projected median S-R as the red dotted line. Top right plot: annual recruitment with errors bars representing 0.95 credible intervals, projection starts in 2019. Bottom left plot: mean fishing mortality, shaded area represents 0.95 credible intervals; red vertical line marks the start of the projection. Bottom right plot: spawning stock biomass (SSB), projected biomass starts in 2019. Orange dashed lines represent reference points estimated from equilibrium analysis and blue dashed lines represent reference points from the simulation method.

Probability of recovery and stock biomass are shown as a function of relative fishing mortality for the different recruitment scenarios (Figure 7.4). For current levels of F (relative F is 1), probability of recovery at B_{msy} level is extremely low in the short-term (less than 10%). It increases slightly for the year 30 of the projection, but still at less than 20% of probability of recovery above B_{msy} values. Even for the best case scenario, if recruitment would double, the probability of recovery at current F values is just above 25% (dark green line in plot). Probability of stock recovery only increases substantially to 50% if fishing mortality would be reduced by half. The overall estimates of SSB are slightly higher for year 30 of the projections. However, SSB at current levels of F is very reduced, and improvements in recruitment do not seem to significantly change the scale of SSB at high levels of fishing mortality.

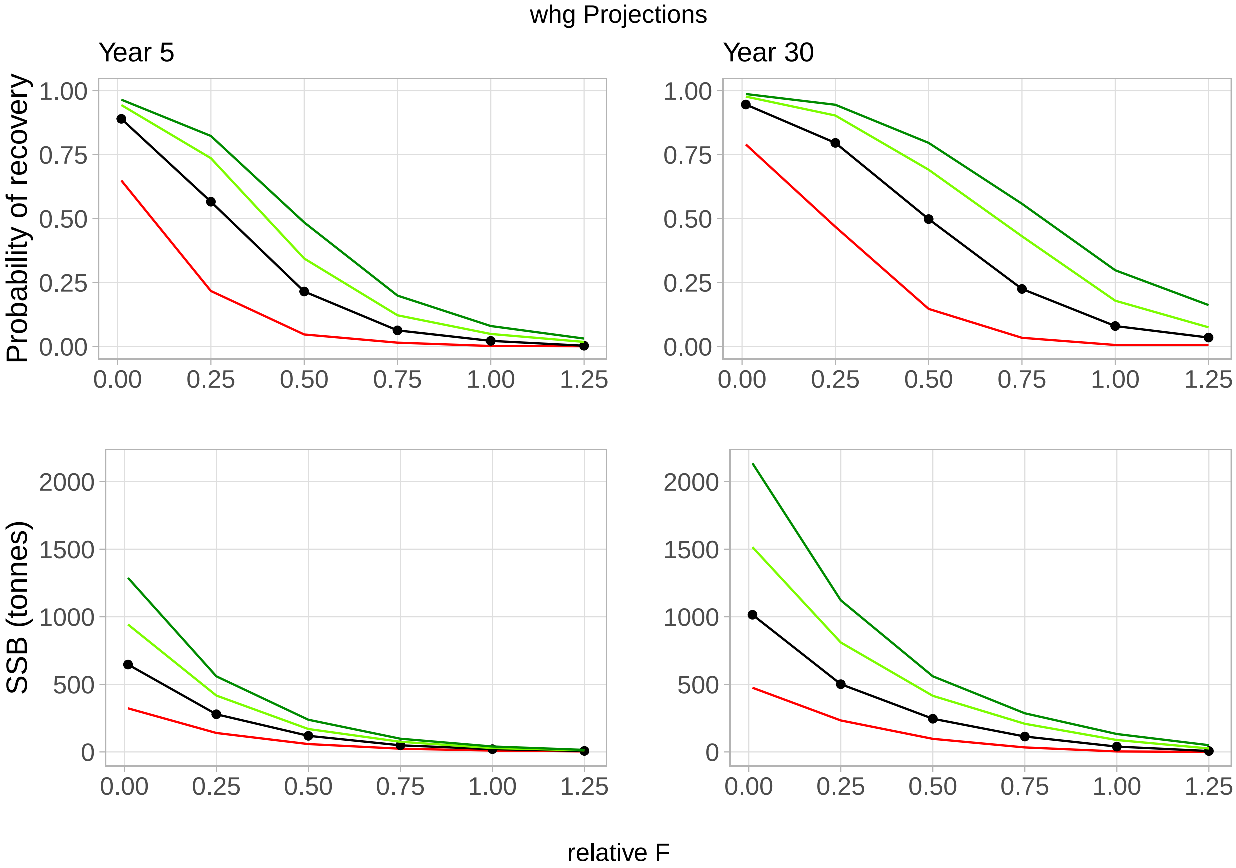

Figure 7.4: Comparison of short-term and long-term projections on the probability of stock recovery and median spawning stock biomass (SSB) for whiting data. Projections were performed at varing levels of fishing mortality and recruitment scenarios. The black dotted line represents baseline recruitment; the red line represents a 0.5 decrease in recruitment; the light green and dark green lines a 1.5 and 2x increase in recruitment, respectively.

7.3.3 Cod projections

The estimated reference points from the equilibrium analysis are shown in Table 7.6. Biomass at MSY (B_{msy}) was estimated at almost 766 tonnes and fishing mortality corresponding to F_{msy} around 0.32.

Table 7.6 - Estimated references points (median and quantiles) from equilibrium analysis for cod data.

Quantile	B_{msy}	F_{msy}	MSY	Rmsy
2.5%	279.05	0.28	81.32	614.82
50%	765.99	0.32	219.87	1577.76
97.5%	3083.67	0.39	824.70	6245.09

Table 7.7 summarizes the reference points that can be extracted from the simulation method. The B_{msy} is considerably lower to the calculated previously in Table 7.6. This can be explained by the change in the peak of the distribution of the B_{msy} values, shown in Appendix F (Figure F.7). B_{msy} from equilibrium analysis (B_{msy}) has a distribution with a long right-hand tail without a clearly defined peak that drives the median to higher values. The B_{msy} distribution estimated from the simulation method still has a long right-hand tail, but the distribution has a clear peak, which might explain the difference between the two estimates. The F_{msy} from the simulation method is similar to the one estimated in Table 7.6.

Table 7.7 - Estimated references points (median and quantiles) from simulation method for cod data.

Quantile	Bmsy	Fmsy	MSY	Rmsy
2.5%	78.92	0.14	73.06	192.19
50%	379.56	0.31	255.90	1592.54
97.5%	2508.46	0.60	1268.61	12779.62

The population was projected for 30 years using the baseline year of 2019. One projection is shown in Figure 7.5, together with the estimates from the ASM between 1985 and 2019. Assuming a stock-recruitment relationship that follows the BH curve, the population is projected to stabilize at values higher than 50 tonnes of SSB and 1000 recruits per year. The left hand side of the population trajectory (red line) corresponds to start of the projection. The projection shows stable annual recruitment values, with wide credible intervals. The population was projected at a constant fishing mortality after 2019, which is substantially higher than F_{msy}. The projected SSB stabilizes at much lower values compared with the B_{msy} estimated from both methods (orange and blue dashed lines).

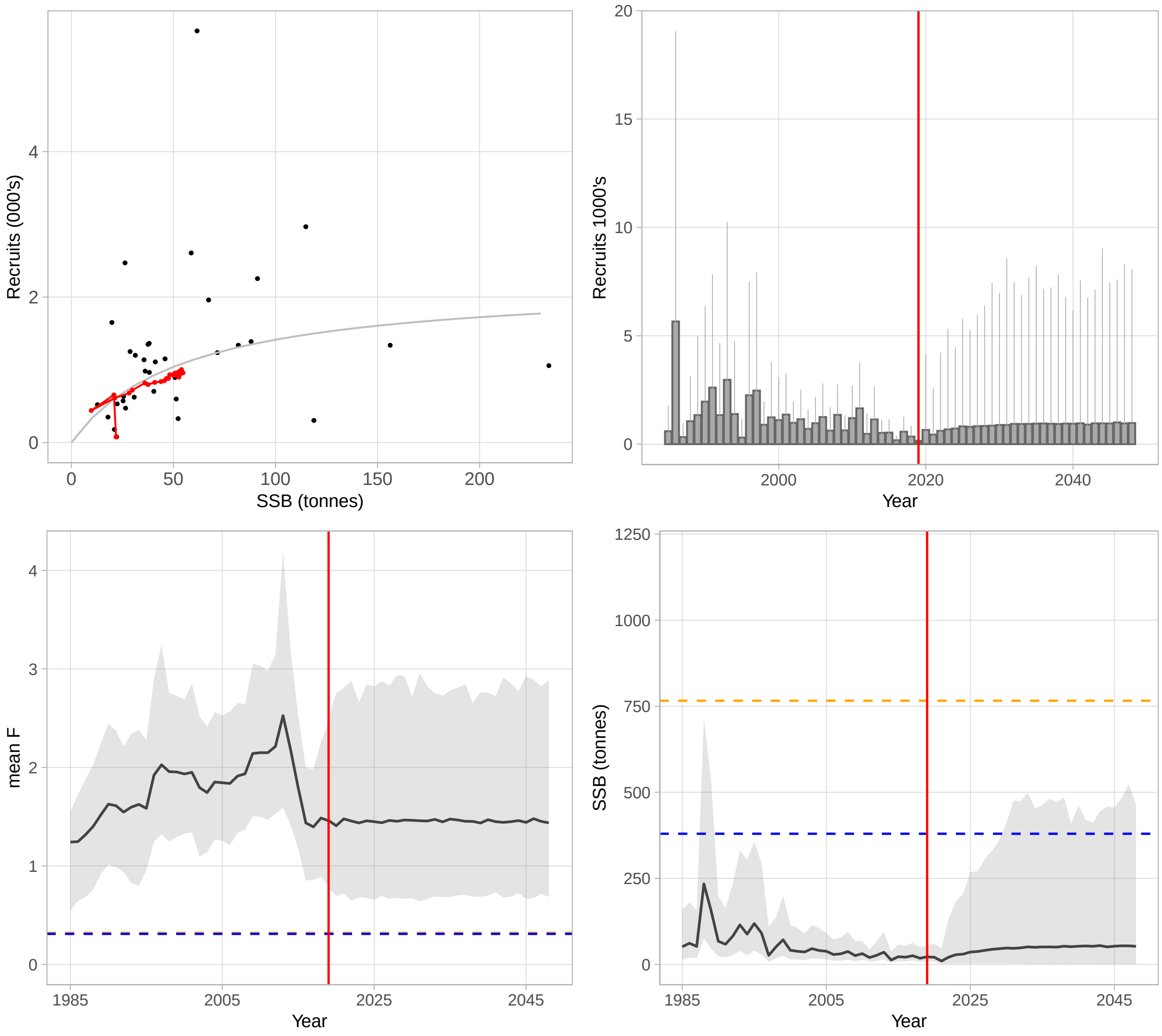

Figure 7.5: Outputs of the projected cod population at current fishing mortality levels, combined with population estimates from 1985 to 2019. Top left plot: stock-recruitment relationship, black dots representing estimates between 1985-2019 and associated Beverton-Holt curve, and projected median S-R as the red dotted line. Top right plot: annual recruitment with errors bars representing 0.95 credible intervals, projection starts in 2019. Bottom left plot: mean fishing mortality, shaded area represents 0.95 credible intervals; red vertical line marks the start of the projection. Bottom right plot: spawning stock biomass (SSB), projected biomass starts in 2019. Orange dashed lines represent reference points estimated from equilibrium analysis and blue dashed lines represent reference points from the simulation method.

Probability of recovery and stock biomass are shown as a function of relative fishing mortality for the different recruitment scenarios (Figure 7.6). For current levels of F (when relative F is

1), probability of recovery is extremely low both in the short-term and long-term projection (less than 10% chance of recovery above B_{msy} levels). Even with a large reduction of 50% in fishing mortality and an optimistic scenario of doubling recruitment levels, the chances of recovery are just above 50% for the long-term projection. In the hypothetical scenario of turning off fishing activities (relative F is 0), probability of recovery is lower than 1, likely because of the large variability in recruitment values. The overall estimates of SSB are slightly higher for year 30 of the projections. However, SSB at current levels of F is very reduced, and improvements in recruitment do not seem to significantly change the scale of SSB.

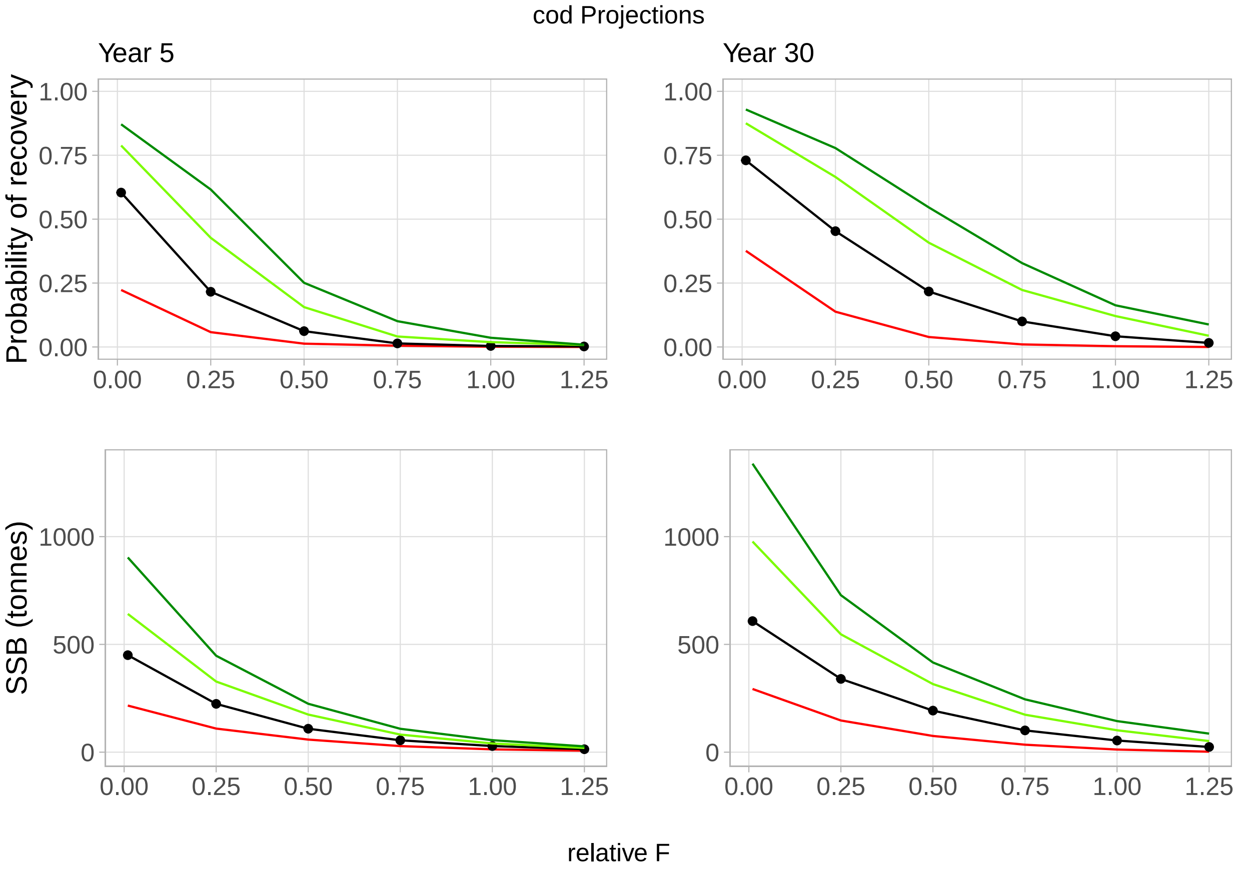

Figure 7.6: Comparison of short-term and long-term projections on the probability of stock recovery and median spawning stock biomass (SSB) for cod data. Projections were performed at varing levels of fishing mortality and recruitment scenarios. The black dotted line represents baseline recruitment; the red line represents a 0.5 decrease in recruitment; the light green and dark green lines a 1.5 and 2x increase in recruitment, respectively.

7.3.4 Comparison between species

The reference points for the three species are shown in Figure 7.7. These were calculated from the simulation method. The biomass at MSY (B_{msy}) for cod contains wide credible intervals compared with whiting and haddock. This can be explained by the skewed distribution of B_{msy} with a long right hand tail shown in the Figure F.7 (Appendix F). Fishing mortality corresponding to MSY (F_{msy}) for cod has the lowest values (around 0.3), for whiting the highest (0.9) and haddock in between with a F_{msy} value of 0.5. Note that recruitment values at MSY for cod are extremely low (less than 2000 recruits per year). This is due to the Beverton-Holt curve reaching an asymptote at lower values of recruitment than for haddock and whiting (see Figure F.8 in Appendix F).

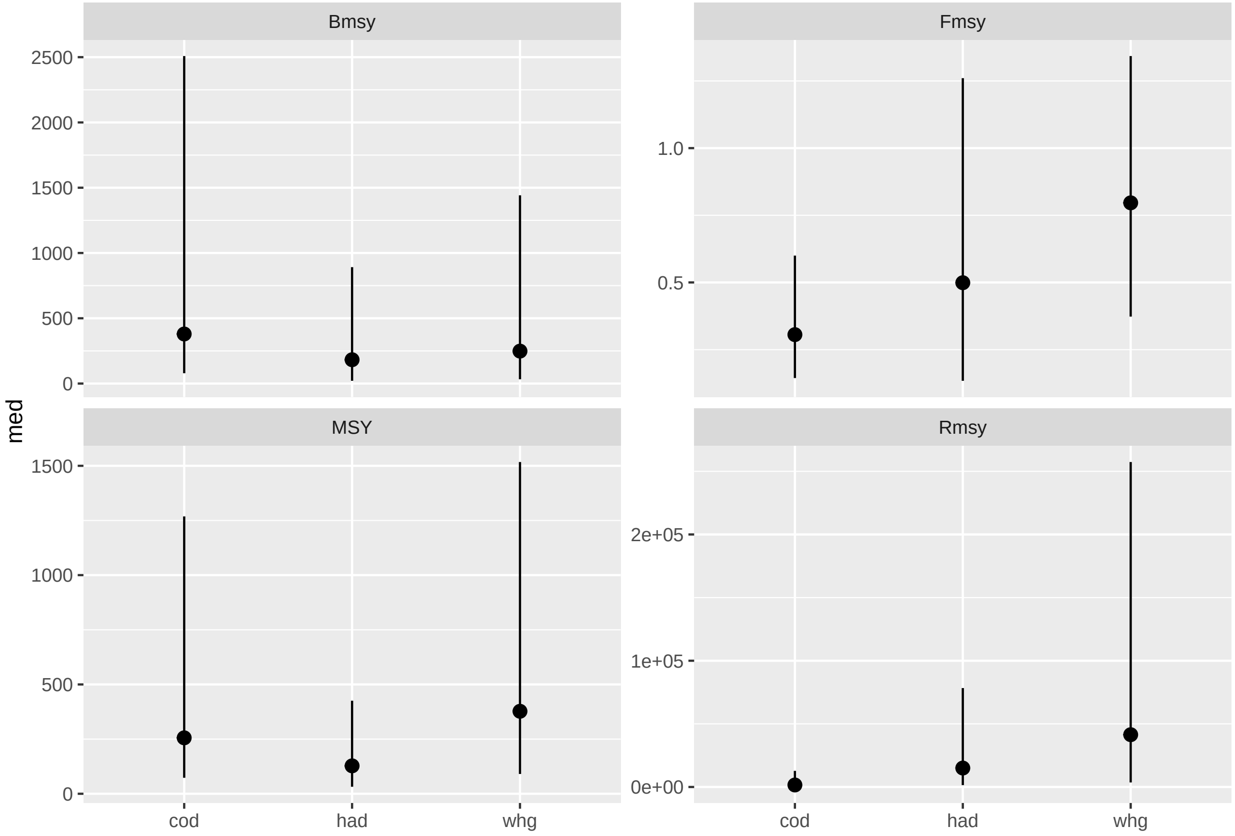

Figure 7.7: Estimated reference points for projected populations at F_{msy}. Points correspond to median measurements, and the segments to the 0.95 credible interval.

7.4 Discussion

A projection model based in outputs from previous Chapters allowed testing of different scenarios of stock recovery for haddock, whiting and cod populations in the Clyde. Biomass reference points, which serve as a reference point for comparison with current biomass levels, were estimated using two methods: equilibrium analysis (directly from the stock-recruitment curve), and by a simulation method that projected the population until equilibrium to find the fishing mortality value that corresponded to MSY. Stock recovery was assessed by calculating the probability of the projected biomass being above the level of B_{msy} estimated from the simulation method. To note that the reference points and probabilities of recovery were estimated under the assumption of independent observations included in the ASM. Given the issue of correlated discard weights explained in Chapter 4, these estimates and their credible intervals represent minimum estimates of uncertainty.

For haddock, the most recent estimate of fishing mortality for 2019 is below F_{msy}, and biomass is higher than B_{msy}. This should allow some stock recovery provided the stock-recruitment relationship holds in the future. However, this is a questionable assumption given that the Clyde haddock is known to be connected to adjacent stocks. Nevertheless, the recent high recruitment values contributed to an increase of spawning biomass in 2019 that will persist in the short-term. Other haddock stocks are described as having sporadic recruitment events, with strong year classes that sustain the stock for several years (Castano-Primo et al., 2014). For the whiting population, current fishing mortality is above F_{msy}, even though F_{msy} is estimated at a higher value than expected (values of F_{msy} around 0.2-0.4 are expected as estimated for the whiting stock in west of Scotland; ICES, 2022b). The current spawning biomass is well below B_{msy}. The probability of the whiting stock to recover above B_{msy} within the next 30 years at current fishing levels is low, less than 20%. Even with high recruitment, this probability would only rise to around 25%. As for cod, the current fishing mortality is substantially higher than F_{msy}, and the current stock biomass is extremely low compared with the B_{msy} levels. Chances of stock recovery are extremely low (less than 10%), even in a scenario of environmental conditions that would favor a recruitment increase of 100%. The probability of recovery for cod is above 0.8 without fishing activities (considering relative

F as 0) and at high recruitment scenarios. This might be due to the fact that recruitment values are highly variable, and there is a chance of obtaining low recruitment regardless of the biomass and fishing mortality levels.

When comparing the results across species, the rate of decrease of probability of recovery with increasing levels of fishing mortality is more pronounced for whiting than haddock. This is probably because the current levels of whiting biomass are very low in the Clyde. The value of F_{msy} for cod is considerably lower than for haddock and whiting. The haddock and whiting stocks still produce high recruitment values at low biomass, while cod has very low recruitment values even at high biomass. This can be seen by comparing the stock-recruitment curves between species (Figure F.8 in Supplementary information). The higher the steepness of the stock-recruitment curve, the more resilient the stock is to high exploitation rates. For cod, the slope of the stock-recruitment curve near the origin is very low and is therefore less resilient to high fishing mortality. The fishing mortality rate is projected at a constant value for the 30 years of the projections. As a result, the stock biomass values are less variable in the projections than the historical values because the fishing mortality is assumed constant, and the mean weights at age and natural mortality are also assumed constant (using 2019 values).

7.5 Conclusions

Overall, the haddock biomass is recovering at current fishing levels (around 50% chances of recovery) and may continue to recover in the short-term future. However, there are reduced chances of recovery for the whiting and cod stocks in the Clyde at current fishing levels. In 2019, the fishing mortality was lower than the immediately preceding years, and these earlier high values will have suppressed the most recent value of stock biomass. The values of fishing mortality are much higher than the reference points, so these need to be reduced to allow stock recovery.

CHAPTER 8 - Discussion and Conclusions

8.1 Overview

The Firth of Clyde demersal fish community has undergone major changes throughout the 20th century, from a state of biomass dominated by top predators such as spurdog and cod in 1960's to only a few species (e.g. whiting and haddock) of small sized individuals by the end of the century (Heath and Speirs, 2012). In the past, there had been various fisheries for pelagic, demersal fish and shellfish in the Clyde. A centuries-long fishery for herring collapsed in the 1970's, and targeted fishing for demersal fish stopped entirely in early 2000's. At the moment, the only fisheries that are economically viable are demersal trawling and creeling for prawns and dredging for scallops to a smaller extent. Multiple hypotheses have been proposed to try to explain the changes in the demersal fish community in the Clyde, these have been described in Chapter 1. This study focused on the hypothesis that fish discards from the Nephrops trawlers in the Clyde might be suppressing the recovery of the whitefish populations (haddock, whiting, cod), acting as a hidden but high mortality rate on these stocks.

To address this question, data was collated, fish population models were developed and applied to each Clyde stock to estimate fishing mortality and stock biomass. This was the first time that a fully age-structured stock assessment model was undertaken for the Clyde demersal fish populations of haddock, whiting and cod.

All fisheries-dependent (observer and logbook data) and fisheries-independent (survey) data for the three species had to be extracted and compiled as a starting point. This was previously described in detail in Chapter 2. The data compiled in Chapter 2 was used in all the subsequent Chapters. For Chapter 3, total fish discards for the Clyde Nephrops trawl fishery were estimated using different approaches, and a novel model-based methodology is proposed to estimate discards that is distinct from previous studies (Stratoudakis et al., 1999; Stratoudakis et al., 2001). In Chapters 4 and 5, an age-based fish population model was successfully developed, tested, and applied to the Clyde stocks to estimate fishing mortality and stock biomass. We used an alternative modelling approach in Chapter 6 where a surplus production model was developed and applied to the Clyde stocks, and the biomass estimates from the surplus production model were compared to the biomass estimates given by the age-structured model. Biological reference points were calculated for the Clyde stocks in Chapter 7, and the populations were projected for the next 30 years under different fishing regimes to quantify the probability of recovery above these reference points.

8.2 Impact of discards in the recovery of the whitefish stocks in the Clyde

Fishing mortality derived from the Nephrops fishery appears to be an important factor hindering the recovery of the whiting and cod stocks in the Clyde. Despite the successful efforts from the industry to reduce bycatch of these species, this still represents a source of mortality that is sufficiently high to have an impact on these stocks. The pessimistic view of low abundance of the whiting and cod stocks in the Clyde are reflected in adjacent areas from the ICES assessments, while the better state of a recovering haddock population in the Clyde has also similarities to the west of Scotland and Irish Sea.

The assessment results for haddock in the Clyde show a recent increasing trend in stock biomass that is likely to be related with peaks in recruitment in recent years. The increase in stock biomass and decline in fishing mortality has also been estimated for the Irish Sea, and for the west of Scotland to some extent (ICES, 2013; ICES 2021b). This means there might be some degree of connectivity of the three stocks, providing an indication that the Clyde haddock population might not be fully isolated from adjacent stocks. The most recent estimate of fishing mortality for 2019 is below F_{msy}, and biomass higher than B_{msy}. Maintaining the current fishing mortality level should allow a biomass increase in the short-term, with probability of recovery above $Bmsy$ level around 60%. At present, the haddock population in the Northern Shelf and in the Irish Sea are also considered within safe biological boundaries, with fishing pressure below F_{msy} and spawning stock size above limit reference points (ICES, 2022a; ICES, 2022b; ICES, 2021b).

For the whiting stock, biomass has been declining since early 2000's, with no indication of recovery. Fishing mortality values are very high, although they have decreased moderately after 2010. When comparing biomass and recruitment patterns over time with adjacent stocks, there does not appear to be any resemblance. Current reference points are outside safe biological limits. Consequently, the probability of stock recovery at current fishing levels is unlikely (less than 30%), even with a high recruitment year class entering the population. This has some similarities with the whiting stock in the Irish Sea. ICES recommends zero catch advice in the Irish Sea, with spawning biomass below reference points and fishing mortality still above F_{msy}. For the west of Scotland, a minor catch quota has been advised since stock biomass has recovered slightly, and fishing mortality has been below F_{msy} since 2005.

Cod stock biomass and recruitment in the Clyde have been declining for the past 34 years and are now at historical low levels. Fishing mortality rate is very high. While cod stock biomass in the Clyde has not shown any signs of recovery, in both the Irish Sea and west of Scotland it seems to have increased slightly after 2010. This is consistent with studies that suggest that cod stocks in the west of Scotland are structured in relatively closed subpopulations (Wright et al., 2006a; Wright et al., 2006b; Gibb et al., 2007; Galley et al., 2006). The chances of cod stock recovery in the Clyde are minimal (less than 10%), even in a scenario that would double the recruitment levels. This pessimistic view of the Clyde stock matches the ICES advice for cod in the Irish Sea and the west of Scotland. The spawning biomass is well below reference points, and fishing mortality is above F_{msy} even though there had been decreases in fishing mortality since 2009. The size of the cod stock in both areas is extremely low, and zero catch advice was given in the assessments up to 2022. However, the current ICES assessment (from 2023 onward) groups the west of Scotland and North Sea stocks as a single unit, and gives a picture of recovering spawning stock biomass above the lower limit but fishing mortality still above F_{msy} (ICES, 2023b). The status of the 6a cod stock is highly uncertain; minor changes in the modelling assumptions can have major implications for the current values of stock biomass (Cook, 2019b).

Overall, at current levels of fishing mortality from the Nephrops fishery operating in the Clyde, the chances of recovery for the whiting and cod stocks are minimal. For the haddock stock, the recent increase in recruitment and decrease in fishing mortality might help the stock to recover in the short-term. The Nephrops fishery catches mostly immature fish (age groups 0- to 2-year-olds). Particularly for cod that only reaches maturity after 2 years, to give the stock any chance of recovery, mortality on these age groups needs to decrease so as to allow current year classes to survive and reproduce, and this would lead to an increase in stock biomass. These results differ from the findings of Alexander et al. (2015). These authors used an ecosystem modelling approach for the whole of the west of Scotland and argue that there is insufficient bycatch by the Nephrops fleet to have a large impact on gadoid stocks. The results of this paper also suggested that moving to a "clean" Nephrops fishery (i.e. without fish landings or fish discards) would result in a decrease for haddock and whiting stocks. This can be explained by the fact that the model treats discards as detritus, and by removing catches/discards, there may be less prey availability for detritivorous species and this can propagate through the food web. Moreover, the analysis was conducted at the level of the west coast of Scotland, but the Clyde region needs to be considered separately from its neighbouring areas.

8.3 Limitations of the analysis

The ASM occasionally estimated improbable changes in historical fishing mortality in years where catch age compositions were not available to use (before 2002). It could be that using only the survey age compositions before 2002 does not provide enough information for properly estimating historical trends in fishing mortality. Nevertheless, only the most recent estimates (for the past 15 years) of fishing mortality and stock biomass were relevant for drawing future scenarios of stock recovery. Another limitation of the data used is that the number of observer trips per year in the Clyde is very low. This will have contributed to higher uncertainty estimates, but the modelling approaches explored in Chapter 2 were able to account for observation error and to greatly reduce the credible intervals. The change in survey design and sampling locations between 2010-2011 meant that the survey data had to be included as four separate surveys, and this reduced the time series for the Q4 post-2011 survey. The relative abundance index estimated for this survey seemed to contain higher uncertainty, but this was dealt to some extent by estimating specific observation error distributions for each survey.

Chapter 4 described the issue of including correlated discard data in the ASM. One solution could be running the ASM using only the design-based estimates as these are unbiased and independent observations. However, the variability of these estimates is very large and would likely destabilize the assessment due to the low level of sampling in the Clyde. An alternative solution would be to include the "raw" discard observations for each trip and estimate the discard weight (or numbers) within the ASM. This could be accomplished by splitting the numbers-at-age into proportions-at-age (i.e. age compositions) and total catch (in numbers) with separate error distributions for each component. A Dirichlet-multinomial error distribution might be used for the proportions, and the lognormal distribution for the total catch (Thorson et al., 2016; Methot and Wetzel, 2013). The use of trip data would overcome the issue of using correlated discard weights.

For all three stocks assessed in the Clyde, fishing mortality was consistently higher than fishing mortality estimated by the ICES assessments for adjacent areas. The age-structured model used in this thesis tracks the total mortality of the cohorts. If the natural mortality values used are too low, then this could lead to an overestimation of fishing mortality. While the partition of total mortality into natural and fishing mortality is debatable to some extent, the values of total mortality are very high for the Clyde stocks, and any possibility of recovery will be dependent on reducing these levels of mortality. Nevertheless, the values of natural mortality were estimated using a similar approach as used by ICES and are in close agreement with these, so there is little evidence to believe that the natural mortality values used are too low. In addition, even if the fishing mortality values would be half the estimated values, these would still be large enough to affect stock recovery as given by the population projections at varying levels of fishing mortality. For cod, fishing mortality at MSY was estimated around 0.32, and this would be achieved with a reduction of 80% in current F. However, the probability of the stock to recover within 30 years is just above 50%. A similar comparison can be made for whiting, where a reduction in F of 40% would reach the Fmsy level, but the probability of recovery is below 50%.

The most likely explanation for the discrepancies between fishing mortality levels estimated for the Clyde and adjacent areas is the high fishing effort density concentrated in the Clyde that drives fishing pressure to a much higher scale. The Clyde is a relatively small area compared to the Irish Sea and west of Scotland, but it can yield a large number of boats fishing continuously throughout the year. As seen in Chapter 5, fishing intensity in the Clyde in days per km^2 can be more than three times higher than fishing intensity in the west of Scotland.

For Chapter 7, we considered the Clyde stocks as isolated populations, where the spawning biomass produces all the recruitment in the Clyde. There is, apparently, evidence of a stock-recruitment relationship, by visual inspection of the stock-recruitment plots and the fitting of the Beverton-Holt curve. However, it is also known that there are migration effects for the whiting and haddock stocks. An unknown proportion of haddock recruits might be produced from adjacent areas, most likely from the West of Scotland. For adult whiting, there is an offshore migration to deeper waters into the west coast, so a proportion of the spawning biomass might not be represented at all in the Clyde. Nevertheless, the ASM follows the cohorts that exist in the Clyde, either arising specifically from the Clyde area or adjacent areas, so the current estimates of biomass correspond to the existent fish biomass in the Clyde.

8.4 Other factors influencing stock decline in the Clyde

Other factors might be in play that can affect the recovery of fish biomass in the Clyde, since reducing fishing mortality to safe levels (i.e. fishing at MSY) does not guarantee a recovery of the whiting and cod stocks.

Climate change is driving species distributions shifts and affecting productivity of the stocks in the North Sea (Perry et al., 2005). Over the past 30 years, there has been an invasion of warmer-water species such as hake and sea bass in the southern North Sea and Celtic Sea, and a retreat of typically cold-water species (e.g. cod, haddock) into deeper parts of UK waters (Heath et al., 2012; Wright et al., 2020; Perry et al., 2005). Cook and Heath (2005) found a negative relationship between temperature and recruitment of North Sea cod stocks, although there is evidence of a positive effect for whiting. Climate change is expected to affect stock status in the long-term (next 30 – 50 years). In the short-term, stock recovery is mostly dependent on reducing fishing effort to allow existing year classes to survive until reaching maturity (Cheung et al., 2012). Moreover, climate change alone cannot be blamed for decreasing stock abundances (Wright et al., 2020; Engelhard et al., 2014). Increasing sea temperatures in both the North Sea and Gulf of Maine lead to contrasting trends of cod stocks over the last three decades; cod stocks increased in the North Sea and declined in the Gulf of Maine (Brander, 2018). Reducing fishing pressure has likely resulted in an increase in stock biomass, although climate change effects such as shifting geographic distributions cannot be fully disregarded. Recent work shows that both fishing and environmental alterations from climate change have equivalent effects on the North Sea cod stock (Beaugrand et al., 2022).

Another factor could be the presence of population depensatory mechanisms, also known as Allee effects. Traditional stock-recruitment dynamics assume that recruits-per-spawner increase with declining stock biomass (compensatory dynamics), an assumption that was included within the stock-recruitment models in Chapter 7. Allee effects imply a positive association between per capita population growth rate and population size at low abundance, meaning that recruits-per-spawner decline with declining stock biomass (Perala and Kuparinen, 2017). There are a variety of reasons that can explain this effect, and they are mostly related with benefits of the presence of conspecifics, e.g. reproductive success depends on the chances of finding potential mates (Stephens and Sutherland, 1999). There is strong evidence of Allee effects in some of the cod stocks in the western Atlantic, based on data on recruitment and stock biomass (Keith and Hutchings, 2012; Perala et al., 2022). These publications suggest that after 30 years of dramatically reducing fishing mortality and numerous fishing closures, depensatory mechanisms might be responsible for preventing the recovery of the Newfoundland cod populations. There is no research done about the potential for Allee effects in the Clyde whitefish stocks, and these are challenging to prove, since data are usually sparse for the range of low stock abundances in which Allee effects would be most noticeable (Stephens and Sutherland, 1999; Myers et al., 1995).

The stock-recruitment relationship depends not only on the stock biomass but is also linked to the growth and condition of an individual fish, since this determines to some extent fecundity, egg size and spawning potential (Subbey et al., 2014). Nevertheless, Wright (2014) showed that survival rate at early life stages is the most determinant factor for recovery potential of gadoid stocks around the world, in comparison with life history indicators such as age at maturity and relative fecundity.

Besides the effects on a single population level, intense fishing in the Clyde seems to have altered substantially the demersal fish community (Heath and Speirs, 2012; Thurstan and Roberts, 2010). If the Clyde ecosystem has suffered dramatic structural and functional changes, it could have entered another regime dominated by small-size fish species and benthic invertebrates that typically occupy lower trophic levels. Nevertheless, overall productivity of the Clyde does not seem to have decreased (Mills et al., 2017; Heath and Speirs, 2012; Lawrence and Fernandes, 2021).

Similar regime shifts have been studied in different ecosystems around the world. In the Black Sea, major shifts between 1970 and 1990 were associated with impacts on the upper levels of the trophic food web. The depletion of top predators by high fishing pressure reduced the trophic system from four levels to three trophic levels and was followed by a substantial reduction of planktivorous fish and a bloom of gelatinous plankton (Daskalov et al., 2007). In eastern Canadian waters (Newfoundland and Gulf of Maine), overfishing lead to the depletion of cod stocks. Even after a fishing moratorium and limited fishing pressure since early 1990s, the cod stocks show no sign of recovery. The fishing effort has been redirected to invertebrate fisheries such as shrimp, lobster and snow crab, that now represent the most valuable species in this area (Hamilton et al., 2004; Howarth et al., 2014). This has remarkable similarities to the Nephrops fishery currently operating in the Clyde. When the structure and functional format of these complex ecosystems is changed through regime shifts, it is not known if these ecosystems can recover to their exact previous states (Daskalov et al., 2007).

8.5 Criticisms of the survey

The Clyde Fisherman's Association (CFA) has suggested that the IBTS survey is not appropriate to sample abundance and distribution of whitefish in the Clyde, since gadoid species have a semi-pelagic behaviour instead of purely demersal behaviour. Following extensive consultation with the CFA and local fishermen, Marine Scotland Science (MSS) conducted an experimental survey in 2014 with a semi-pelagic gear design to investigate these reports of semi-pelagic whitefish in the Clyde (Turrell et al., 2016). The semi-pelagic fishing shows that there are large, older white fish in the deep basins of the Clyde, but these are present in very low densities. Besides, any direct comparison with the IBTS survey could not be made, because the gear design and mesh size used were different, with different trawl durations and sampling different locations in the Clyde. Subsequently, the CFA tried to establish a number of surveys in the Clyde between 2016 and 2018 with a standard Nephrops trawl to monitor changes in abundance and distribution of cod and other gadoid species. However, the implementation of these surveys lacked consistency to permit any comparative analysis (Kingston et al., 2022). The IBTS survey remains the most reliable source on information to estimate relative abundance of whitefish in the Clyde.

8.6 Fisheries management implications

The ICES report in 2022 on identification of cod stocks concluded that there are multiple subpopulations in the west of Scotland (ICES, 2022d). One subpopulation is located in offshore areas connected to the North Sea, and a separate subpopulation is in the Clyde with a possible connection to the Irish Sea. However, a more recent benchmark workshop grouped the west of Scotland and northern North Sea as a single unit, named as Northwestern cod stock (ICES, 2023b). The Firth of Clyde is included as part of this wider unit, even though there is clear evidence that support an isolated cod population in the Clyde. This work provides the scientific basis for conducting a separate assessment for the Clyde cod stock.

Since 2001, the Scottish Government has implemented a Statutory Instrument (SI) that closed a specific area in south of the Clyde to directed fishing for 11 weeks each year (the “cod box” closure; Scottish Statutory Instrument, 2022). This area is known as the Clyde sill, and it represented an important spawning ground where cod would aggregate from wider areas to spawn (Armstrong et al., 2005). Under this regulation, targeted fishing for cod was forbidden with gears such as demersal trawl, seine or any static net. However, trawling for Nephrops and dredging were still allowed in the cod box closure. In 2022, an alteration to this legislative instrument was implemented, where no fishing activity is allowed to take place in the “cod box” to prevent any disturbance in the spawning grounds. This excluded the Nephrops fisheries (both trawlers and other gears) from the “cod box” during the seasonal closure. This change in legislation and how it was implemented caused a great controversy between the government and the fishing industry, and lead to multiple discussions at the Scottish Parliament to improve communication between stakeholders and reach consensus on the measures to be put forward (Scottish Parliament 2022a; Scottish Parliament, 2022b). There is a clear societal and public interest in rebuilding the cod stock in the Clyde, and this work contributes with updated abundance estimates and future projections of the cod population in the Clyde that can be used as a scientific basis to inform fisheries management measures like the “cod box” closure.

A number of technical measures to reduce bycatch have been already implemented in the Clyde Nephrops fishery, and these might explain to some extent the decreases of bycatch quantities of whiting and haddock between 2002 and 2019. However, current mortality on cod and whiting is still too high and needs to decrease on these stocks for any chance of recovery to happen, so other bycatch mitigation strategies could be explored further. Bergmann et al. (2002a) detected differences in discards species composition and total discarded quantities of the Nephrops fishery between the north and south areas of the Clyde. Trawls in the south contained a higher biomass of Nephrops and fish discards, whereas catches from the north contained more invertebrates. More recent research is needed on discard composition and quantities to provide insights if there are seasonal and spatial bycatch hotspots in the Clyde that need to be actively avoided. The “cod box” implemented in the south of Arran has the potential to help the cod stock biomass to recover in the upcoming decades. Another measure of bycatch mitigation that could be investigated is the use of VMS (Vessel Monitoring System) and fishery catch data in real time closures to avoid bycatch hotspots. One example is the recently developed app called “BATmap” (BATmap, 2020). This pilot project allows skippers on the west coast of Scotland to record hauls of unwanted species (e.g. cod and spurdog) in real-time and if catches are higher than an agreed threshold, it sends an automatic alert to other participating vessels so they can avoid these areas. It is likely that this software would need to be adapted to work for the Nephrops Clyde fishery and its implementation would have a number of challenges, but there is potential for cooperation and compliance in live reporting of bycatch by an industry-controlled bycatch minimization strategy.

8.7 Future research

Further work could be invested in digitizing age compositions of discards data collected by on-board observers before 2002, so this data could be integrated in the age-structured model. The ASM model is currently fitted to the numbers-at-age in the surveys and the catch assuming lognormal errors. Where zeros occur, a hurdle model is used to avoid problems with the lognormal assumption. As previously described, an alternative approach would be to separate the numbers-at-age into proportions and total number with separate error distributions for each component (Dirichlet-multinomial and lognormal, respectively). This approach is sometimes used in Stock Synthesis (Methot and Wetzel, 2013).

An interesting approach that could be investigated in the future would be developing an ecosystem model for the Firth of Clyde, that would be able to integrate not only multiple demersal fish species and their interactions but also interactions with predators/prey and different fishing fleets. One example could be using the end-to-end ecosystem model StrathE2E (Heath, 2012), that simulates nitrogen fluxes between all levels of the trophic food web including detritus and inorganic matter. Although an ecosystem model would be a better representation of the system being studied and might allow detection of regime shifts in the Clyde, the scarcity of fisheries-dependent and fisheries-independent data, as well as data on other important parameter inputs, will likely provide a significant challenge.

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Appendix A - Chapter 2

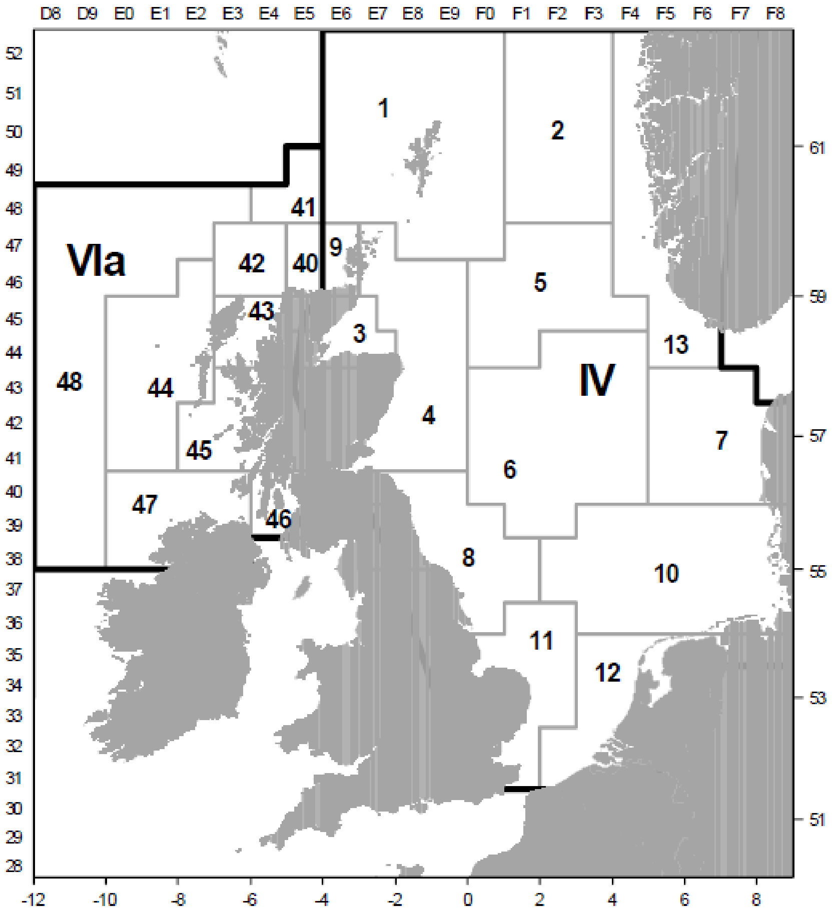


Figure A.1 - West of Scotland sampling areas (from Millar and Fryer, 2006). VIa is ICES division 6a (west of Scotland), IV is ICES subarea 4 (North Sea). ICES statistical rectangles are indicated in the upper horizontal axis and left vertical axis, latitude in lower horizontal axis and longitude in right vertical axis.

Table A.1: Length-weight relationships parameters and conversion factor (gutted weight to total weight).

Parameters	Haddock	Whiting	Cod
a (mean annual)	0.0157	0.0093	0.0175
a (March)	0.0145	0.0090	0.0168
a (June)	0.0151	0.0089	0.0172
a (November)	0.0164	0.0097	0.0182
b	2.8268	2.9456	2.8571
G	1.16	1.13	1.17

Age-length key for observer data

ALKs can be defined as the proportion of fish in an age class a given a certain length l ($P(a|l)$) and were calculated for each trip t as:

$$P_t(a|l) = \frac{n_{t,a,l}}{\sum_{a=1}^{A} n_{t,a,l}} \tag{A.1}$$

where $n_{t,a,l}$ is the number of sampled fish at age and length for each trip.

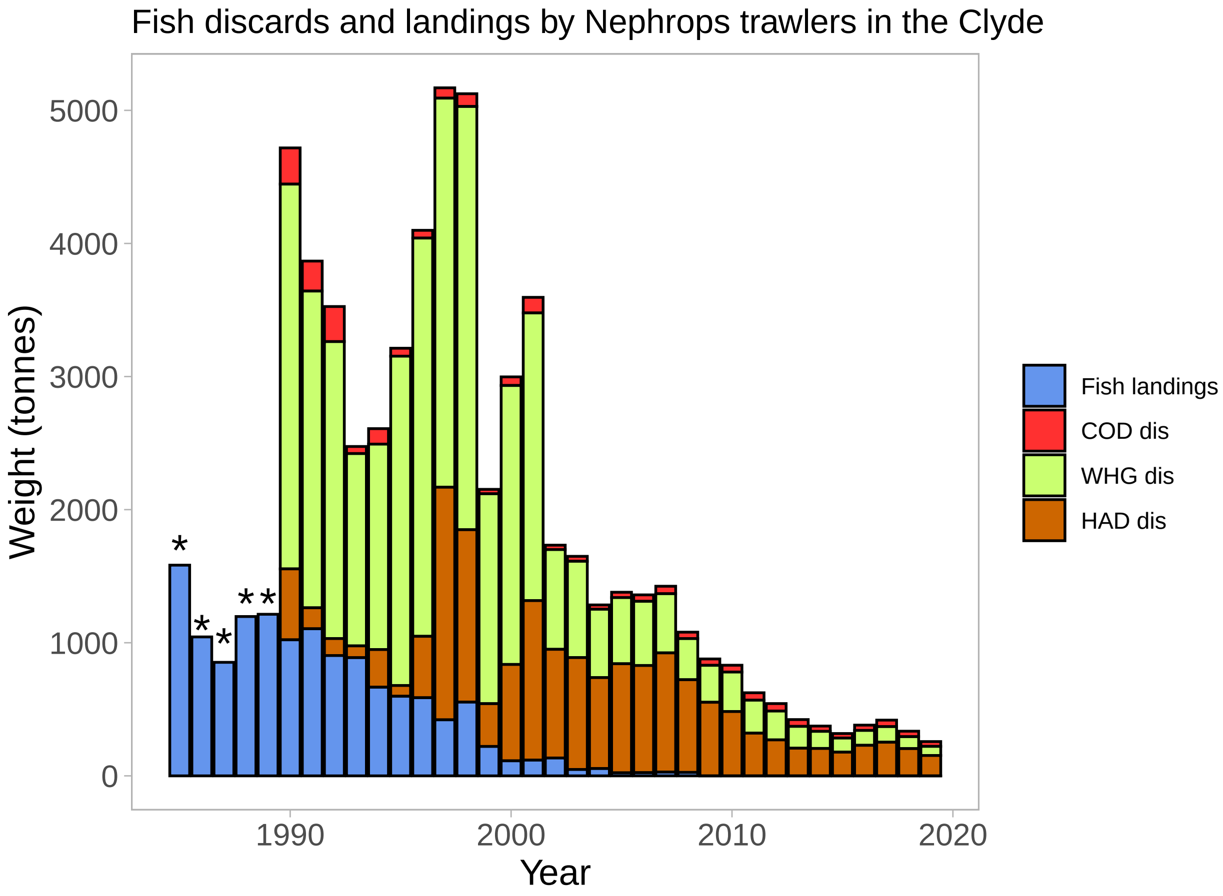


Figure A.2 - Fish discards and landings (cod, haddock and whiting) by Nephrops trawlers in the Clyde between 1985 and 2019. * represents no discards data available.

Nephrops landings as proxy for fishing effort

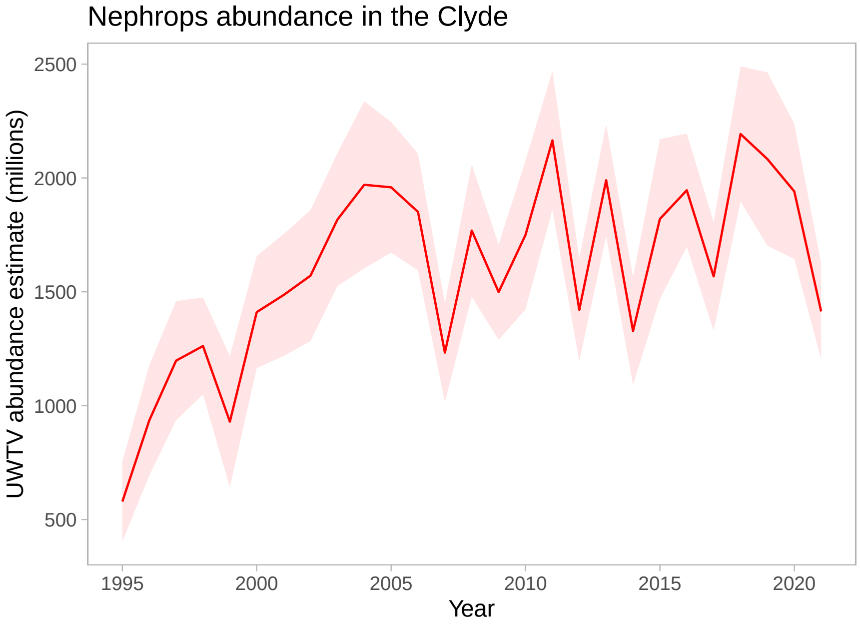


Figure A.3 - Nephrops abundance estimates in the Clyde from underwater TV survey (ICES, 2020a).

NOTE: The Nephrops abundance fluctuates to some extent from year to year, and this will add noise to the estimates of discards because the assumption of constant Nephrops biomass is violated to some degree. However, the yearly changes in discards and fish biomass in the sea will probably be amplified by measurement error, so this will likely not have a large influence.

Age-length key for survey data

To calculate a relative index of fish abundance as numbers per age class, for each species, the proportion of fish at age given length (i.e. age-length key, ALK) was determined using the same equation as A.1 but for each quarter of the year:

$$P_Q(a|l) = \frac{n_{Q,a,l}}{\sum_{a=1}^{A} n_{Q,a,l}} \tag{A.2}$$

where $n_{Q,a,l}$ is the subset of numbers of fish at length l and age a, for either the Q1 or Q4.

To get the numbers of fish at age:

$$n_{Q,a} = \sum_{l=1}^{L} (P_Q(a|l) n_{Q,l}) \tag{A.3}$$

where $n_{Q,l}$ are the length sampled fraction without age subsampling.

Mean weight at age

The proportions of fish at age and length were converted to weight at age and length distributions:

$$W_{y,a,l} = P_y(a|l) W_{y,l} \tag{A.4}$$

with $W_{y,l}$ being the weight of an individual fish at a certain length class (l) in a given year (y). The mean weight at age is calculated as:

$$W_{y,a} = \frac{\sum_{l=1}^{L} W_{y,a,l}}{nl_{y,a}} \tag{A.5}$$

where $nl_{y,a}$ are the number of length classes in each age group and year.

Smoothed mean weight at age

The posterior distribution of the parameters is proportional to the Likelihood term and the prior distributions, given by the Bayes' theorem:

$$\pi(\sigma_a^{obs}, cv, W_{1,a} | W_{y,a}^{'}) \propto L(W_{y,a}^{'} | \sigma_a^{obs}, cv, W_{1,a}) p(\sigma_a^{obs}) p(cv) p(W_{1,a}) \tag{A.10}$$

The likelihood is defined as:

$$L(W_{y,a}^{'} | \sigma_a^{obs}, cv, W_{1,a}) = \prod_{n=1}^{N} normal(W_{y,a}^{'} | \sigma_a^{obs}, cv, W_{1,a}) \tag{A.11}$$

Figure A.4 - Smooth mean weights at age for the Q1 and Q4 survey data for haddock (HAD), whiting (WHG) and cod (COD). Points represent observations, the lines are the model fit. Age groups represented in Q4 survey data should read one year younger (e.g. age 1 should be age 0, age 2 should be age 1, etc.)

Appendix B - Chapter 3

1.0 Design-based mean estimator

The mean estimator is the simplest estimator of total discards per year (D_y), applied to each species, and takes the mean discard weight per sampled trip (total discards weight divided by the number of sampled trips) and multiples it by the total number of fleet trips as:

$$\hat{D}_y = \frac{T_y}{t_y} \sum_{t=1}^{t_y} \hat{d}_{t,y} \tag{B.1}$$

where T is the number of fishing trips by the Nephrops fleet in the Clyde, t is the number of observed trips and $d_{t,y}$ is the quantity of fish discarded in each observed trip. This estimator is simple to calculate and unbiased (Thompson, 2012). The mean discard weight per year is represented by d_y and the variance of the mean estimator is:

$$var(\hat{D}_y) = \frac{T_y(T_y - t_y)}{t_y(t_y - 1)} \sum_{t=1}^{t_y} (\hat{d}_{t,y} - d_y)^2 \tag{B.2}$$

A 95% confidence interval for the total discards per year for the mean estimator can be computed as

$$\hat{D}_y \pm tdist\sqrt{var(\hat{D}_y)} \tag{B.3}$$

where tdist is the upper $\alpha/2$ point of Student's t distribution with t-1 degrees of freedom since the number of sampled trips per year is considered low (n<30; Table 2.1 in Chapter 2). *alpha* represents significance level.

2.0 Design-based ratio estimator

A yearly ratio (r) between discard weight and Nephrops landed weight for the sampled trips is calculated and then raised at fleet level using Nephrops landed weight for the fleet. Total discards per year from the ratio estimator ($\hat{D}_y$) are calculated as:

$$\hat{D}_y = L_y \hat{r_y} = L_y \frac{\sum_{t=1}^{t_y} \hat{d}_{t,y}}{\sum_{t=1}^{t_y} l_{t,y}} \tag{B.4}$$

where L_y is Nephrops landed weights for the whole fleet, $d_{t,y}$ the discard weight per sampled trip, $l_{t,y}$ is the Nephrops landed weight per sampled trip; r_y is the ratio between discard weight and the auxiliary variable. The variance of the ratio estimator is:

$$var(\hat{D}_y) = \frac{T_y(T_y - t_y)}{t_y(t_y - 1)} \sum_{t=1}^{t_y} (\hat{d}_{t,y} - \hat{r_y} l_{t,y})^2 \tag{B.5}$$

An adjustment to the variance was applied to correct for extreme values of the variance calculated for samples with extreme values of mean Nephrops landings, $\bar{l}$ (Thompson, 2012):

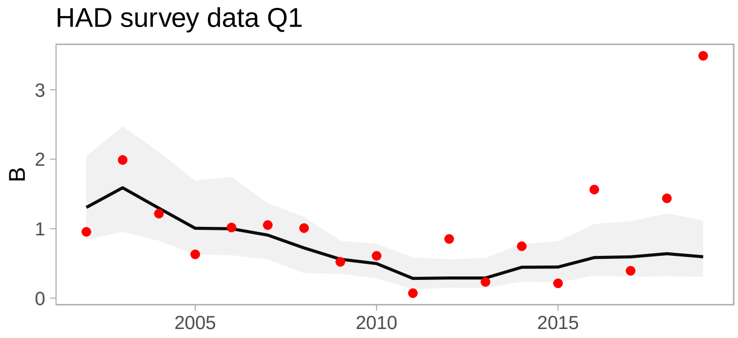
$$v\tilde{a}r(\hat{D}_y) = \frac{L_y/T_y}{\bar{l}_y} var(\hat{D}_y) \tag{B.6}$$

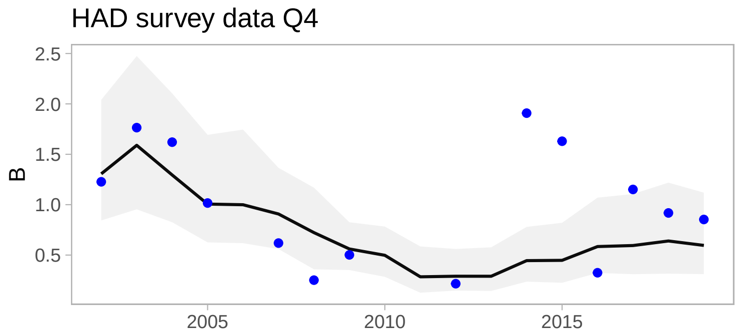
A 95% confidence interval for the total discards per year for the ratio estimator can be computed as

$$\hat{D}_y \pm tdist\sqrt{v\tilde{a}r(\hat{D}_y)} \tag{B.7}$$

where tdist is the upper $\alpha/2$ point of Student's t distribution with t-1 degrees of freedom.

3.0 Combined model


HAD survey data Q1


HAD survey data Q4

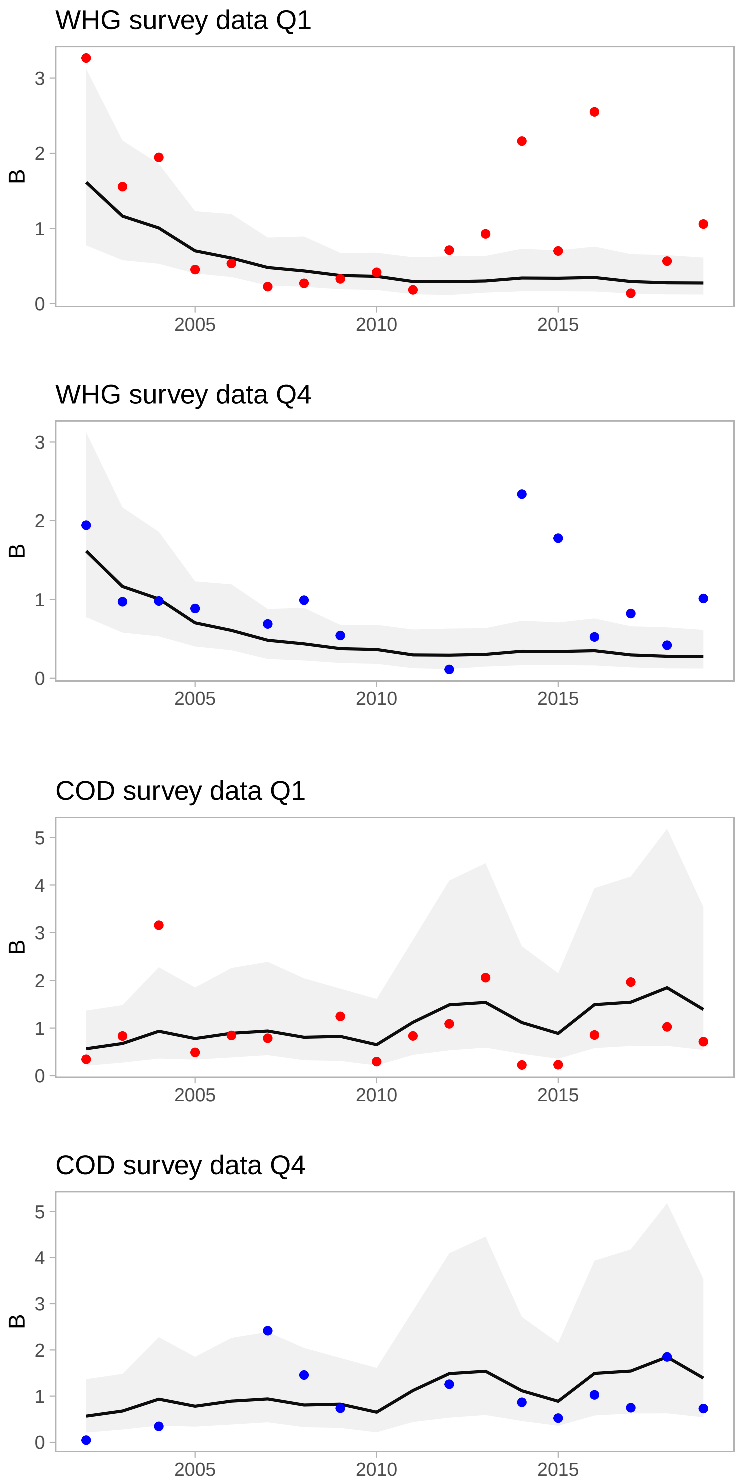


Figure B.2 - Survey index data (dots) and biomass trend (B_y) fitted to the survey data of both quarters (line) for each species. Surveys before and after 2010-2011 are plotted in the same graph, but were considered separately in the analysis.

The change in the survey protocol between 2010 and 2011 means that the catchability parameter (q) will not be the same for the entire time period. Sampling at different stations every year affects the probability of the survey gear to catch fish, as well as modified ground gear that might be more efficient in sweeping the sea floor (ICES, 2017a). Shorter tow duration can also give the opportunity to larger/more fit individuals escape the trawl nets. In addition, separate surveys are conducted within one year (Q1 and Q4 surveys), which in practice means that they sample different fractions of the population since the autumn survey is able to capture the juveniles born between spring and autumn of the same year (0 aged fish). For convenience, each survey index was standardized to the series mean prior to the analysis. This allows to account for complications on the use of the survey data and to extract a common abundance trend from the four time series of surveys.

Calculation of DIC, pD and R-squared

Defining a_i as any observation, θ any model parameter and L the log-likelihood function, the effective number of model parameters (pD) can be calculated as (Spiegelhalter et al., 2002):

$$pD = \mathbb{E}_{\theta|a_i}[2\ L(a_i|\theta)] - 2\ L(a_i|\tilde{\theta}(a_i)) \tag{B.8}$$

where $L(a_i|\theta)$ is the likelihood for each model sample, $\mathbb{E}_{\theta|a_i}$ is the mean across samples, and $L(a_i|\tilde{\theta}(a_i))$ is the likelihood evaluated at the mean of the posterior distributions of the parameters (i.e. mean of fitted values).

The deviance information criteria (DIC) is calculated as:

$$DIC = 2\ L(a_i|\tilde{\theta}(a_i)) + 2pD \tag{B.9}$$

The DIC can only be used to compared models that use the exactly same amount of data. The R-squared (coefficient of determination) based on the geometric mean (log transformed data) was therefore used to compare models that were based on different data sources.

The R-squared is calculated as:

$$R^2 = 1 - \frac{\sum(\bar{a}_{obs} - \bar{a}_{fit})^2}{var(\bar{a}_{obs})} \tag{B.10}$$

where $\bar{a}_{obs}$ is the geometric mean calculated for the data, and $\bar{a}_{fit}$ the geometric mean of the fitted values:

$$\bar{a}_{obs} = exp(\frac{1}{n}\sum_{i=1}^{n} log(a_i)) \tag{B.11}$$

$$\bar{a}_{fit} = exp(\frac{1}{n}\sum_{i=1}^{n} log(\hat{a}_i)) \tag{B.12}$$

Fleet discard numbers at age to be used in subsequent Chapters

We used the discard estimates derived from the mean estimator with a time series smoother (equations 3.1 - 3.4). To convert the discard weights to numbers at age:

i) The numbers of fish at age for each trip ($n_{t,a,l}$) were summed across trips to get numbers of discarded fish at age for each year ($n_{y,a}$)
ii) To raise the numbers at age at fleet level, a raising factor for each year (rf_y) was calculated as:

$$rf_y = \frac{\hat{D}_y}{\sum_{t=1}^{T} d_{t,y}} \tag{B.13}$$

where $\hat{D}_y$ is the fleet discard weight per year. The sampled weight per trip ($d_{t,y}$) was summed across trips for each year to get the sampled discard weight per year (discard weight from the observer trips). Multiplying the numbers of fish at age by the raising factor, we obtain discard numbers at age for the Nephrops fishery, represented as catch numbers at age ($\hat{C}_{a,y}$):

$$\hat{C}_{a,y} = rf_y n_{y,a} \tag{B.14}$$

Appendix C - Chapter 4

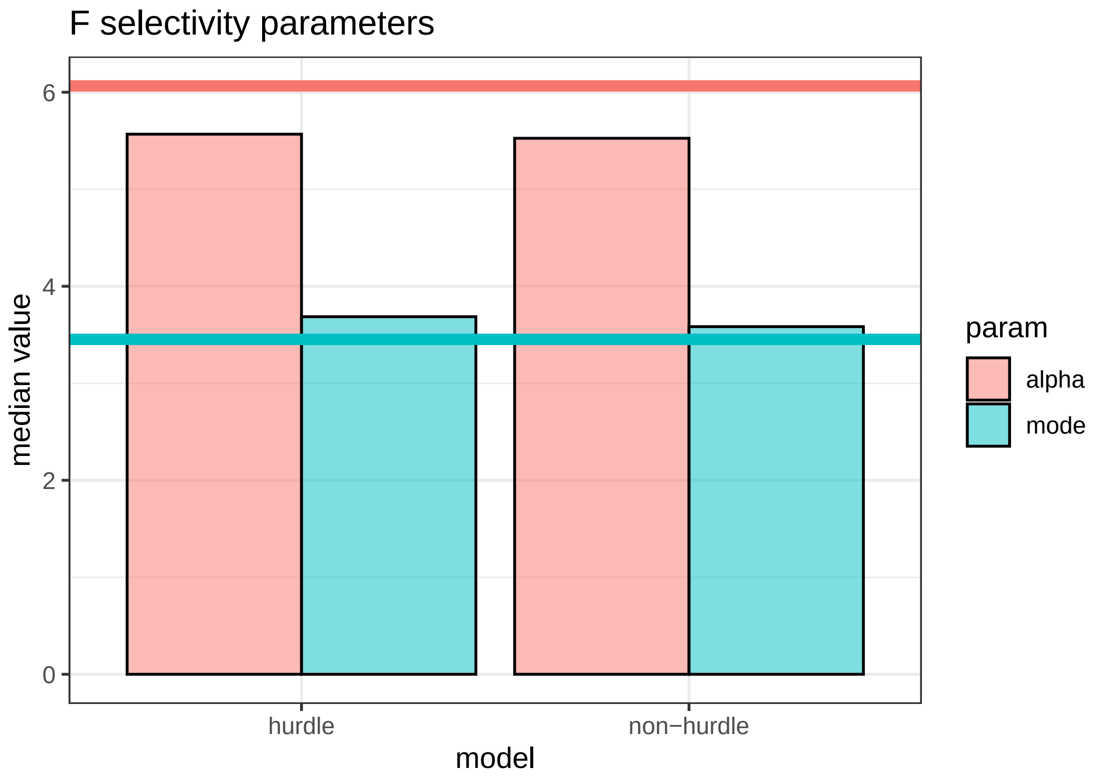


Figure C.1 - Fishery selectivity parameters (alpha and mode) from the simulated data estimated for the hurdle and non-hurdle models. The bars show the median value across the 30 simulations. The horizontal lines represent the true values.

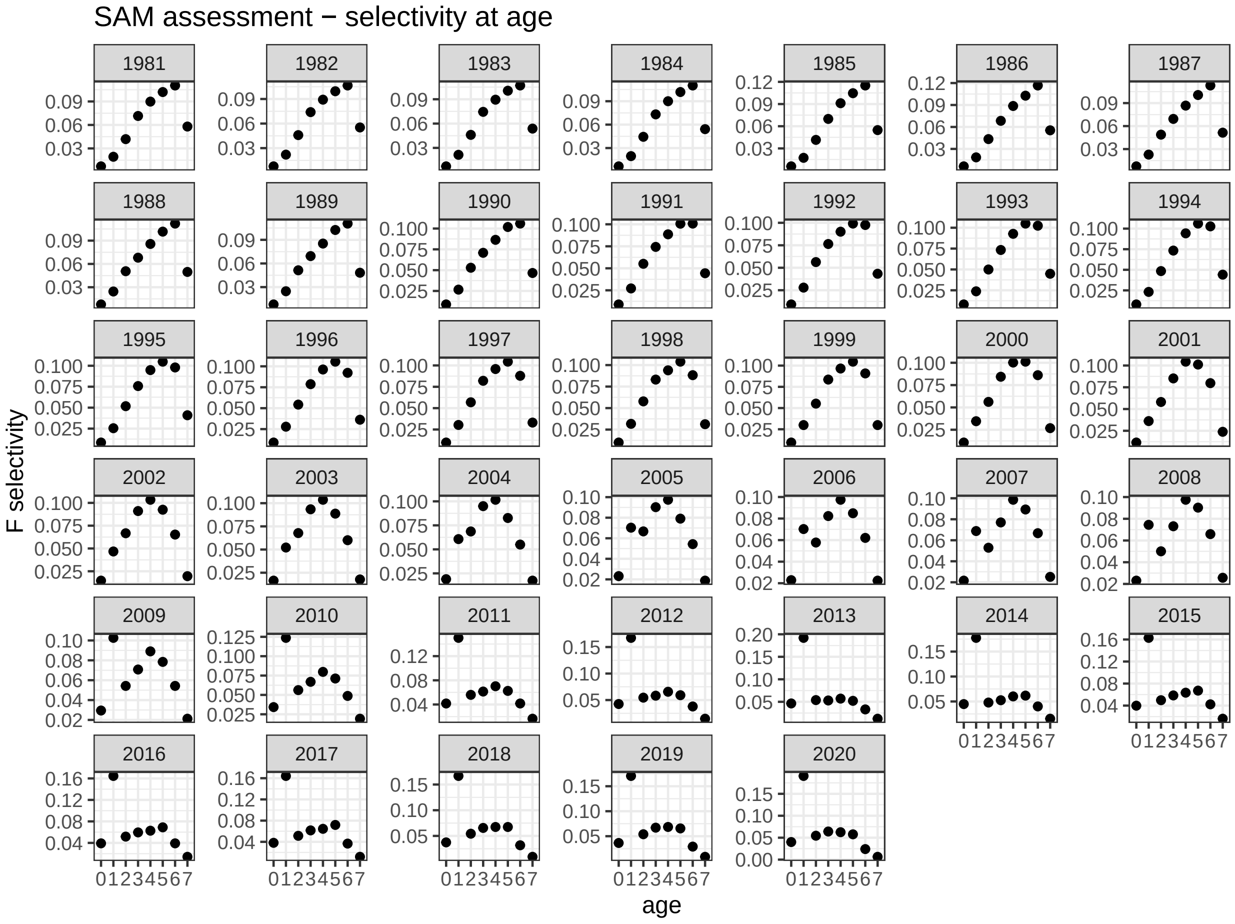


Figure C.2 - Selectivity at age for the SAM assessment (whiting 6a data).

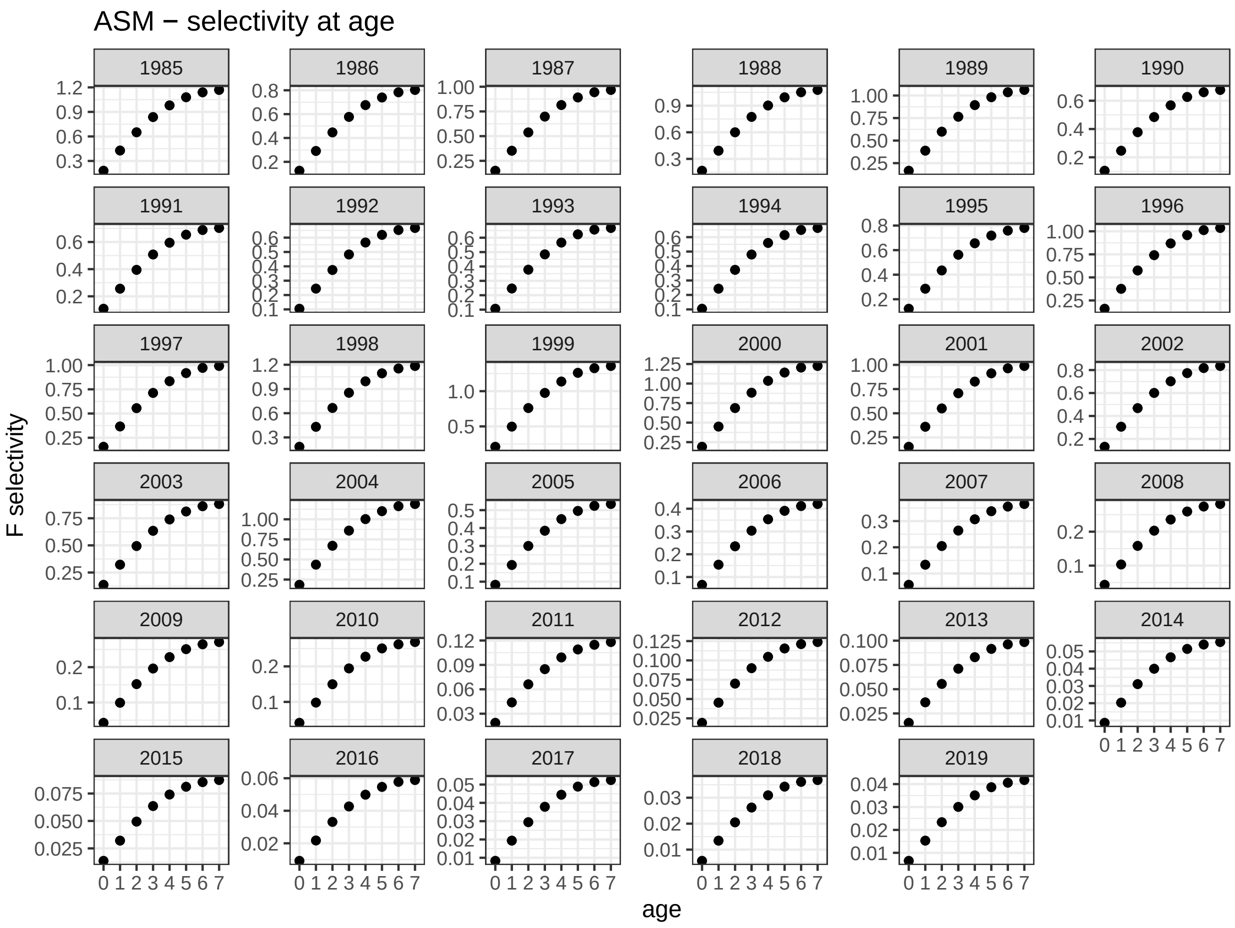


Figure C.3 - Selectivity at age for the ASM model estimated from ICES data (whiting 6a data).

Appendix D - Chapter 5

ASM applied to haddock data

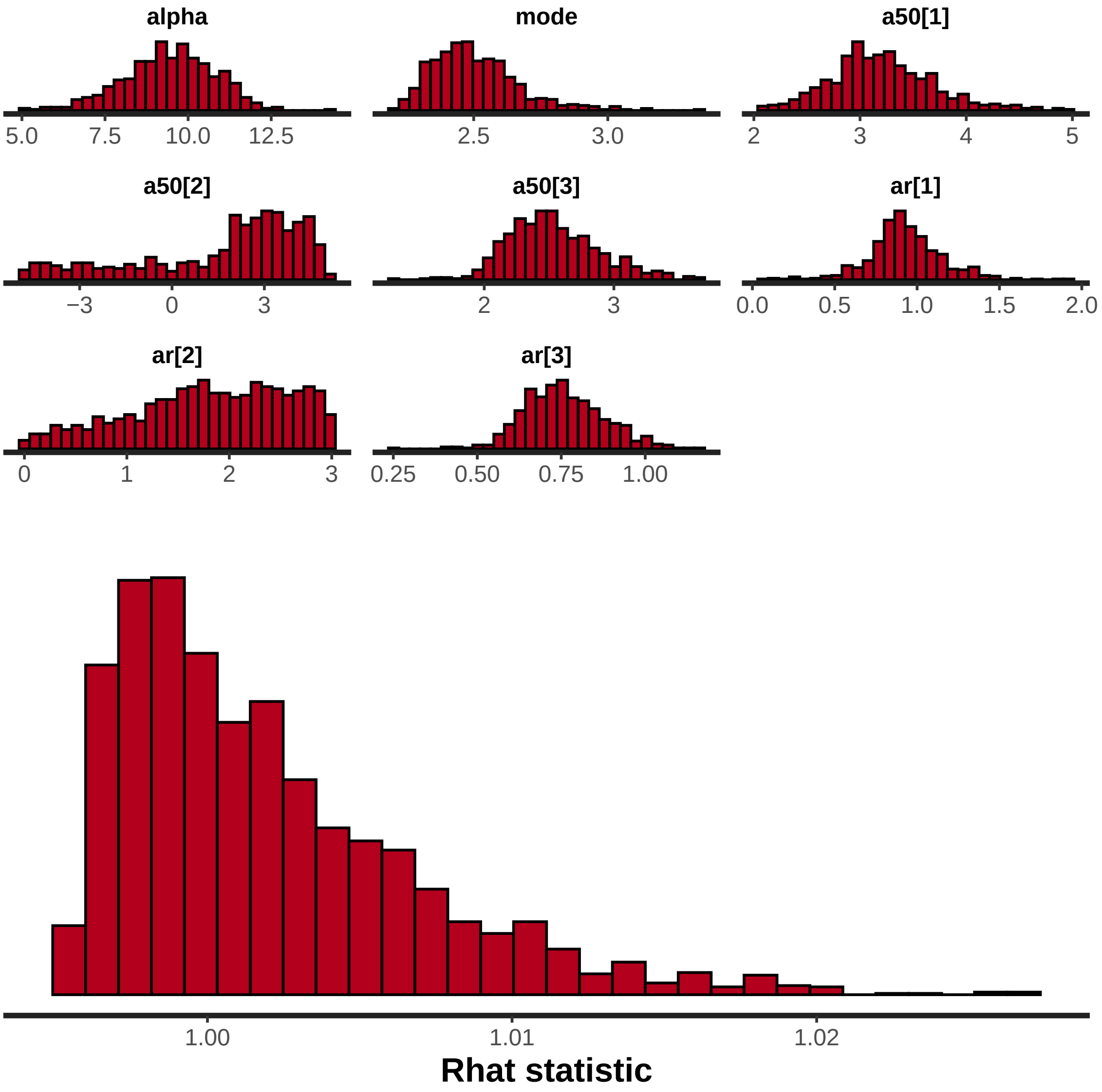
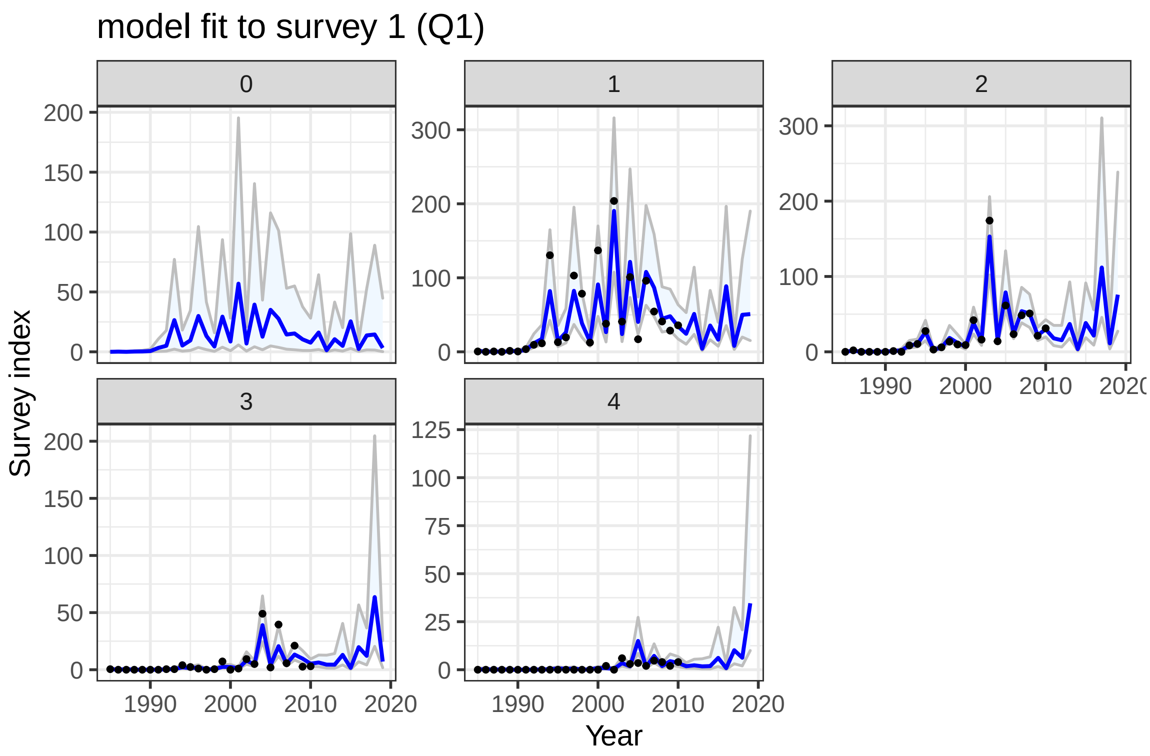
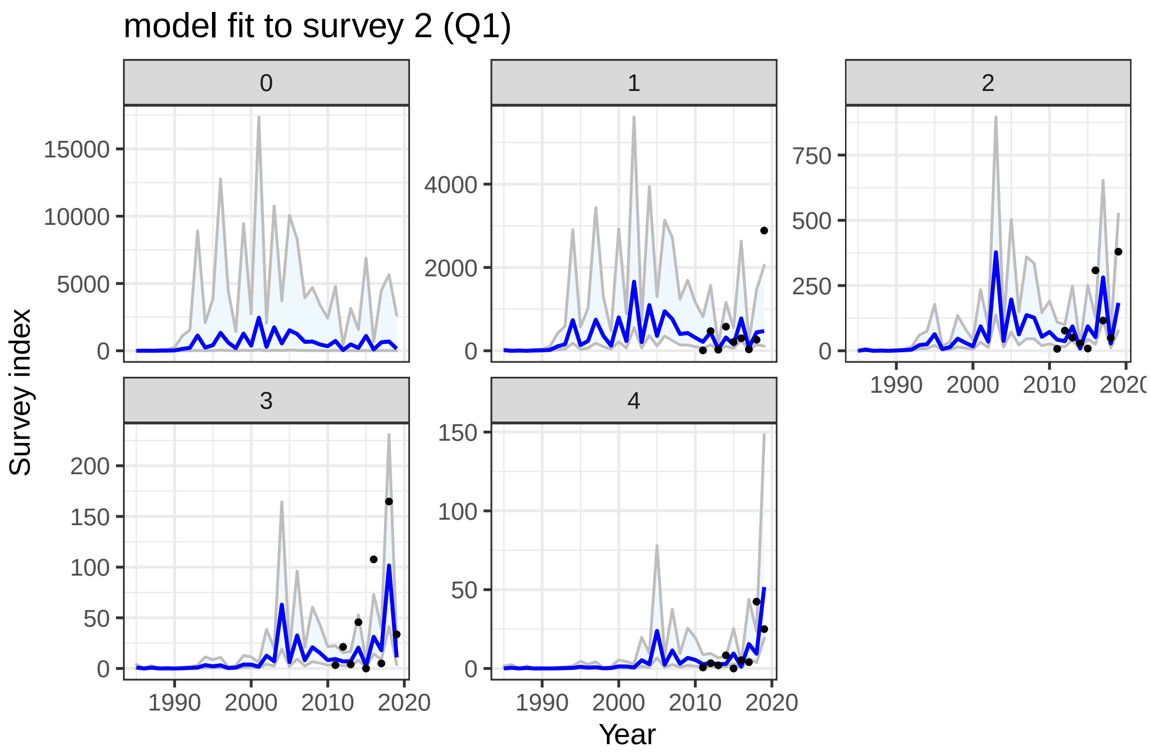
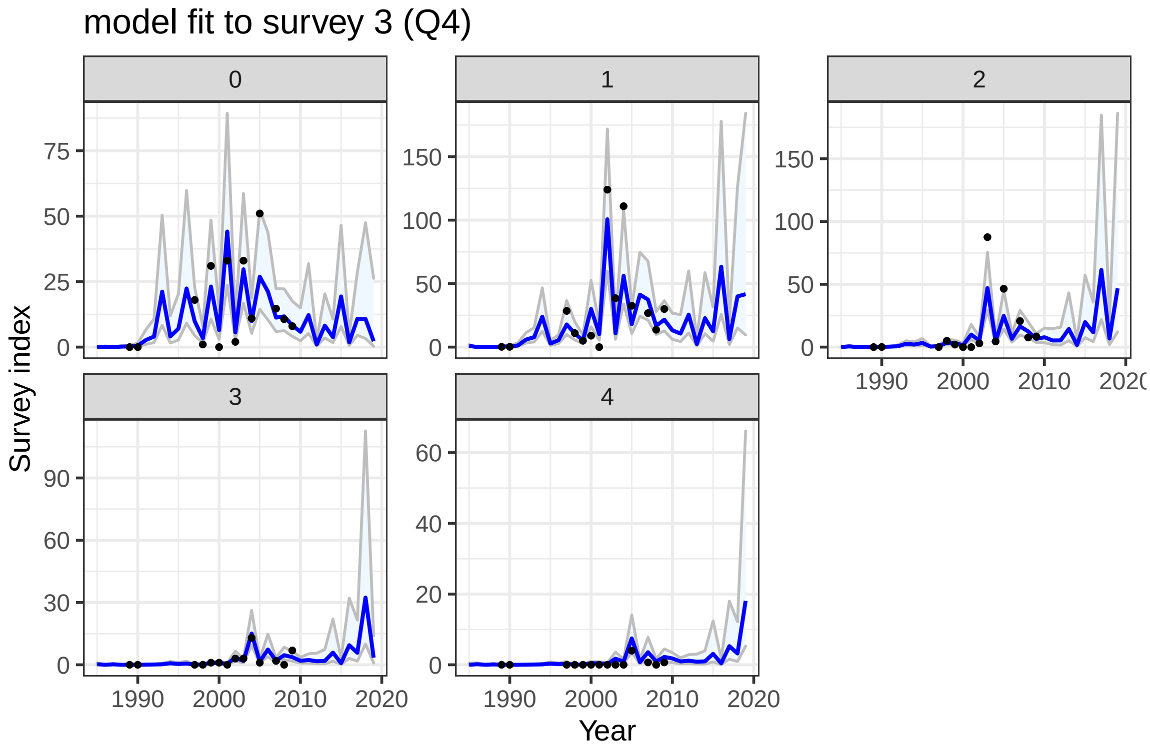


Figure D.1 - Histograms of critical parameters and Rhat value for the reference model applied to haddock Clyde data.


model fit to survey 1 (Q1)
0
1
2
3
4
Survey index
Year
200
150
100
50
0
300
200
100
0
300
200
100
0
125
100
75
50
25
0
1990
2000
2010
2020


model fit to survey 2 (Q1)
0
1
2
3
4
Survey index
Year
15000
10000
5000
0
4000
2000
0
750
500
250
0
200
150
100
50
0
150
100
50
0
1990
2000
2010
2020



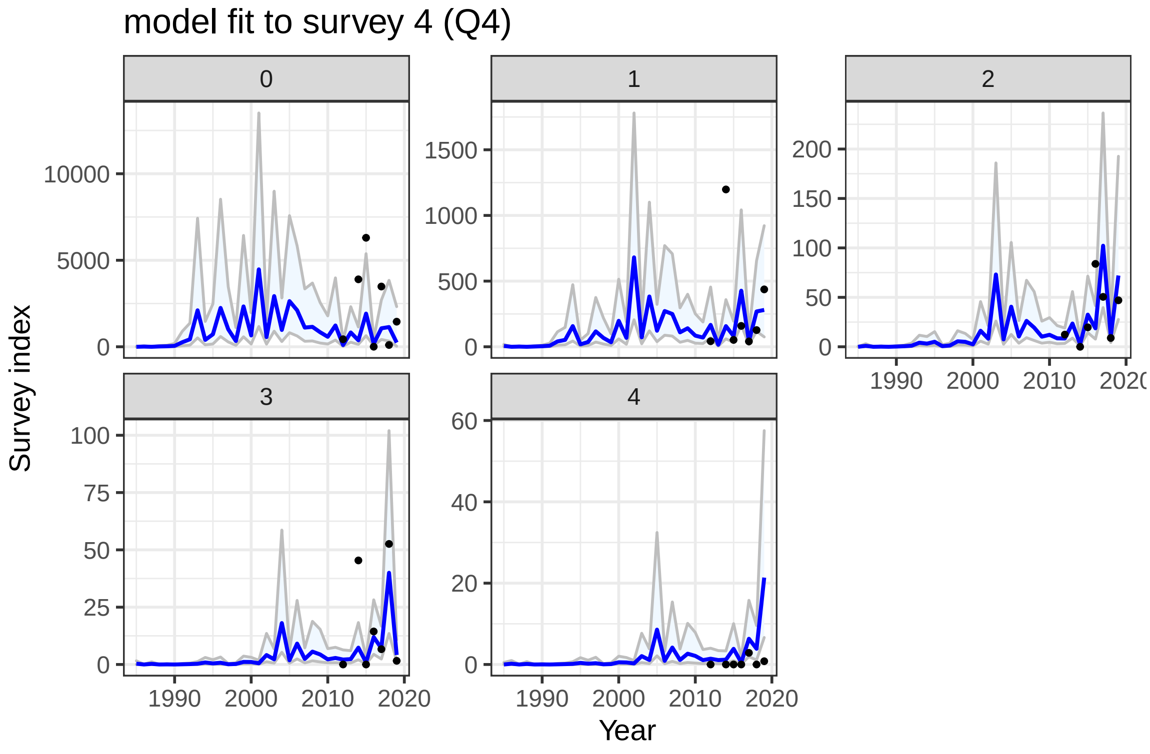


Figure D.2 - Model fit to survey data for the reference model applied to haddock Clyde data for ages 0-4.

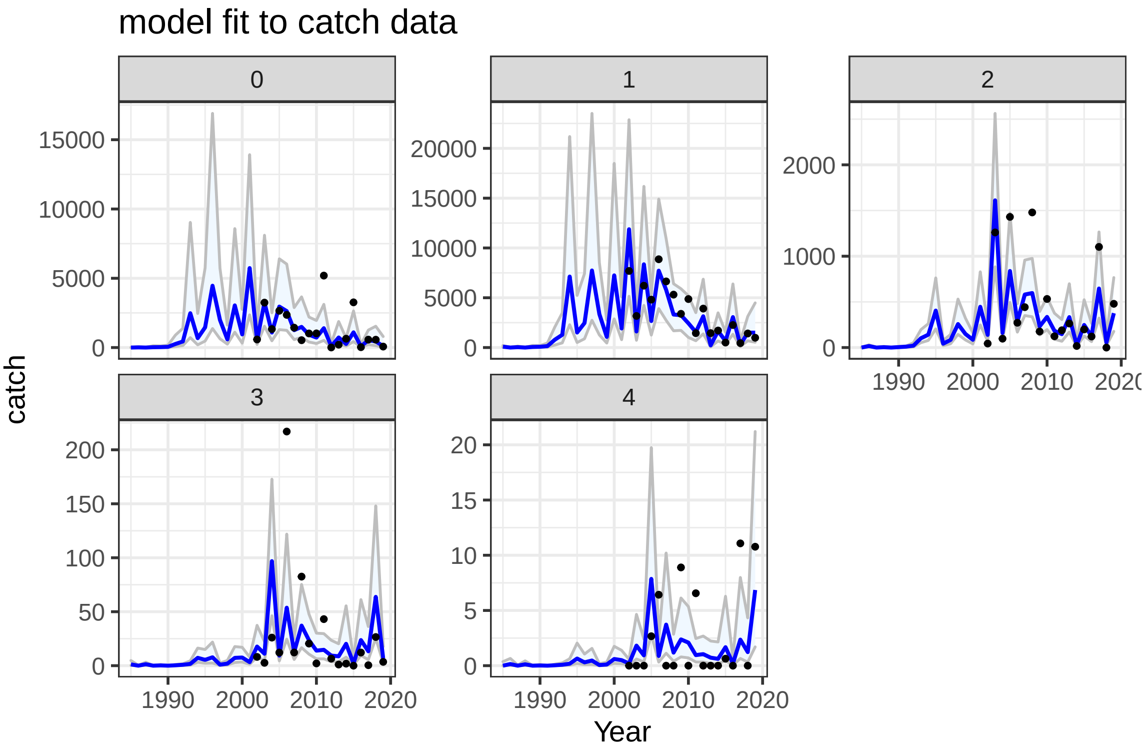


Figure D.3 - Model fit to catch data for the reference model applied to haddock Clyde data for ages 0-4.

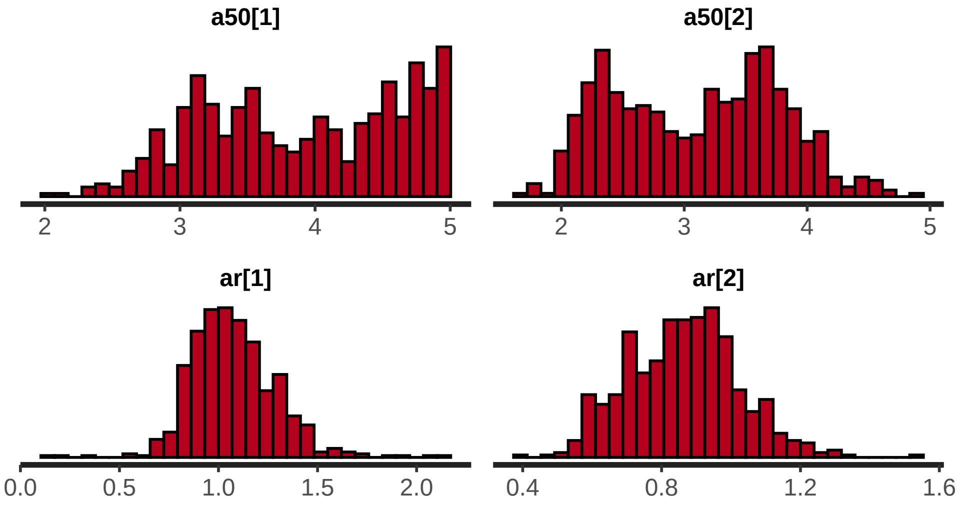


Figure D.4 - Bi-modal distribution of survey selectivity parameters for model run B applied to haddock Clyde data.

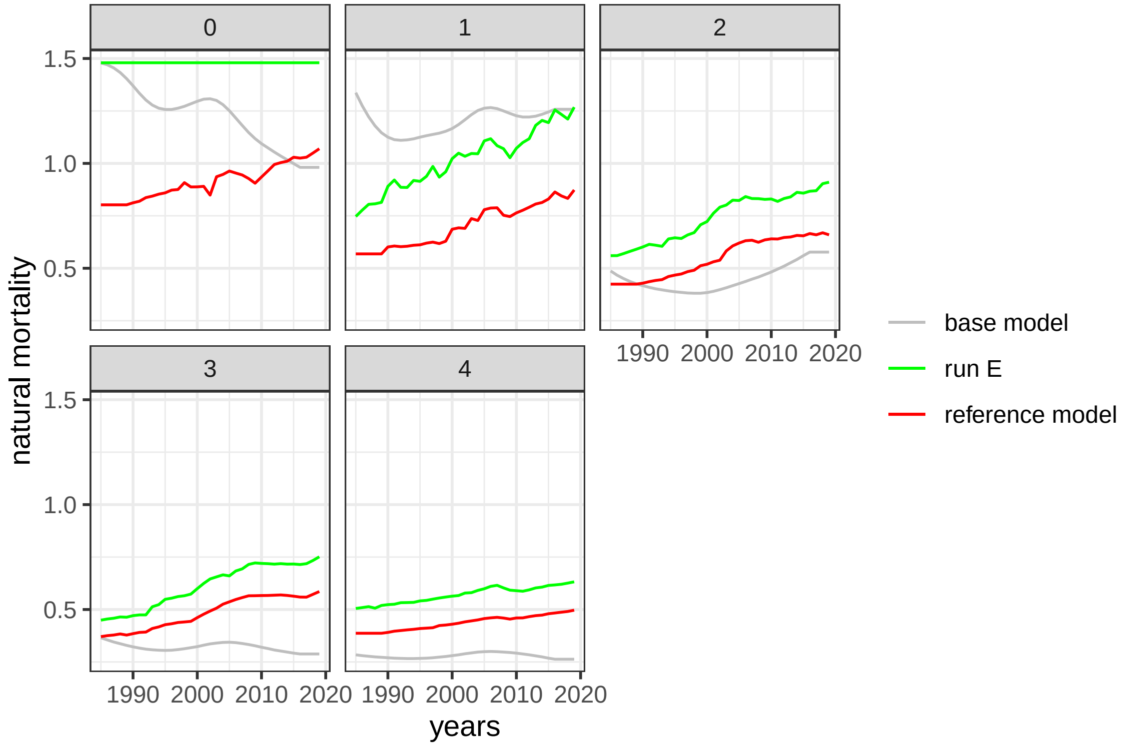


Figure D.5 - Comparison of natural mortality at age values used for the base model, model run E and reference model applied to haddock Clyde data.

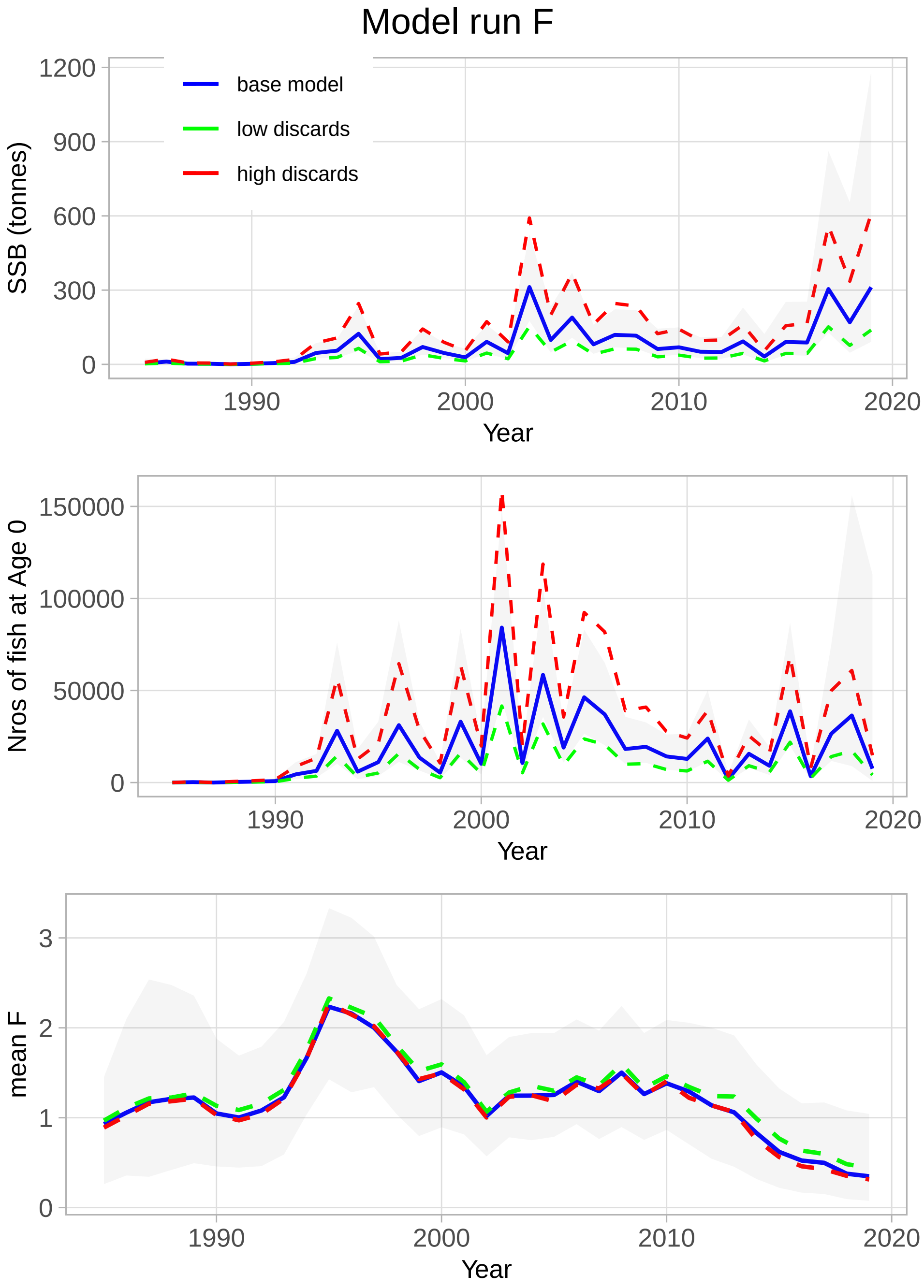


Figure D.6 - Summary statistics and selectivity curves for the model run F applied to haddock Clyde data compared with the base model. Red and green dashed lines represent model fits to upper and lower discards data, respectively. The blue line is the base model. Shaded areas represent 0.95 credible intervals. Mean F calculated over ages 1 to 3.

Table D.1 - Stock summary statistics for each model run/configuration using haddock data. B represents spawning stock biomass (in tonnes), F is mean fishing mortality, and R is a measure of recruitment (number of fish at age 0 in thousands). Base stands for base model, run Z represents the reference model run.

years	base_B	base_F	base_R	runA_B	runA_F	runA_R	runB_B	runB_F	runB_R	runC_B
1985	3.96	0.93	0.00	NA	NA	NA	2.48	1.40	0.00	3.30
1986	11.02	1.06	0.24	NA	NA	NA	8.63	1.55	0.21	10.81
1987	2.83	1.17	0.00	NA	NA	NA	1.87	1.70	0.00	2.32
1988	2.61	1.21	0.31	NA	NA	NA	1.55	1.76	0.26	2.48
1989	0.50	1.22	0.50	NA	NA	NA	0.28	1.79	0.52	0.41
1990	2.30	1.05	0.80	NA	NA	NA	1.68	1.66	0.71	2.37
1991	5.15	1.00	4.47	NA	NA	NA	4.19	1.53	4.62	5.27
1992	9.93	1.08	6.36	NA	NA	NA	7.06	1.65	6.70	10.13
1993	45.91	1.23	28.16	NA	NA	NA	39.03	1.83	25.54	48.49
1994	55.44	1.66	5.92	NA	NA	NA	46.16	2.29	4.54	55.24
1995	123.77	2.23	11.09	NA	NA	NA	104.33	2.78	9.04	135.32
1996	21.33	2.16	31.20	NA	NA	NA	17.72	2.77	27.15	20.35
1997	26.19	2.00	13.59	NA	NA	NA	20.71	2.72	12.28	26.77
1998	70.19	1.73	5.39	NA	NA	NA	62.89	2.30	4.58	78.46
1999	45.89	1.41	33.08	NA	NA	NA	37.99	1.96	25.12	48.74
2000	28.12	1.50	10.18	NA	NA	NA	21.78	1.93	9.05	26.83
2001	91.25	1.35	84.21	NA	NA	NA	77.49	1.81	83.14	100.10
2002	44.99	1.02	10.63	53.45	0.85	13.57	36.87	1.52	8.81	41.27
2003	312.46	1.24	58.52	380.32	1.15	66.02	276.00	1.56	53.07	312.86
2004	98.20	1.25	18.97	138.38	1.07	21.18	86.14	1.71	17.05	87.06
2005	189.22	1.25	46.34	243.37	1.08	54.49	142.88	1.65	41.60	179.16
2006	80.87	1.40	37.02	107.37	1.29	42.55	65.38	1.86	32.12	77.95
2007	119.26	1.30	18.23	150.22	1.11	22.96	99.45	1.74	15.21	118.97
2008	115.93	1.50	19.42	142.87	1.43	24.09	93.79	1.92	17.30	111.19
2009	61.92	1.26	14.19	79.90	1.14	13.52	46.98	1.66	11.38	58.45
2010	68.93	1.38	12.86	87.73	1.60	11.31	59.22	1.76	12.22	67.35
2011	51.01	1.29	23.94	39.04	1.18	25.20	42.06	1.76	23.42	42.07
2012	50.02	1.14	1.82	42.26	1.07	2.38	40.73	1.67	1.62	34.86
2013	92.92	1.06	15.63	89.98	1.09	18.42	76.83	1.61	15.02	66.30
2014	31.57	0.83	9.11	30.80	0.68	8.17	18.12	1.45	8.62	18.88
2015	90.47	0.62	38.72	109.00	0.49	46.74	61.47	1.14	36.15	62.36
2016	88.19	0.52	3.48	96.75	0.39	4.32	55.56	1.08	3.08	46.45
2017	304.49	0.50	26.57	390.91	0.37	31.47	194.93	1.03	22.89	163.26
2018	169.78	0.38	36.45	242.00	0.25	35.24	73.51	0.91	27.09	66.67

2019	311.82	0.35	7.56	463.31	0.23	9.60	148.66	0.83	4.55	127.50

years	runC__F	runC__R	runD__B	runD__F	runD__R	runE__B	runE__F	runE__R	runZ__B	runZ__F	runZ__R
1985	1.08	0.00	1.79	1.30	0.00	14.75	0.54	0.00	4.41	1.04	0.00
1986	1.26	0.30	8.22	1.46	0.21	26.57	0.62	0.32	14.02	1.22	0.15
1987	1.39	0.00	1.66	1.59	0.00	10.32	0.72	0.00	3.43	1.38	0.00
1988	1.45	0.42	1.83	1.69	0.25	8.38	0.77	0.46	3.50	1.46	0.23
1989	1.55	0.71	0.38	1.78	0.49	2.05	0.78	0.71	0.61	1.47	0.33
1990	1.34	1.07	1.79	1.64	0.68	5.91	0.57	1.38	3.06	1.23	0.62
1991	1.25	5.93	4.19	1.53	3.83	12.40	0.56	6.80	6.47	1.17	3.22
1992	1.32	8.50	7.23	1.60	5.58	25.67	0.61	10.07	13.81	1.26	5.06
1993	1.51	37.76	38.22	1.83	21.36	100.50	0.74	51.48	58.02	1.41	25.93
1994	1.97	7.62	44.49	2.27	3.91	128.99	1.17	11.47	67.89	1.94	5.33
1995	2.56	14.51	107.27	2.87	7.60	260.57	1.64	19.70	157.48	2.41	9.67
1996	2.47	40.30	15.94	2.91	22.36	62.91	1.48	67.62	24.59	2.23	30.98
1997	2.36	19.45	21.16	2.69	10.49	61.01	1.49	28.55	32.65	2.23	13.76
1998	2.07	7.10	65.23	2.31	4.14	145.36	1.24	9.00	91.04	1.96	4.66
1999	1.73	44.11	37.93	2.03	24.32	104.41	0.86	54.59	56.35	1.57	29.85
2000	1.81	13.41	19.48	1.99	7.71	76.11	0.93	16.27	34.40	1.69	8.56
2001	1.67	99.90	80.08	1.90	71.88	197.46	0.72	114.42	118.36	1.52	58.57
2002	1.30	12.40	31.40	1.60	9.21	125.86	0.47	14.78	54.07	1.15	6.88
2003	1.42	68.13	250.75	1.72	49.69	691.69	0.65	79.82	388.06	1.29	39.69
2004	1.48	21.77	63.25	1.83	15.85	312.15	0.61	27.79	114.75	1.29	13.24
2005	1.42	55.71	134.09	1.76	39.75	485.41	0.68	73.48	228.62	1.27	36.64
2006	1.65	43.44	59.24	1.97	30.47	213.42	0.80	56.98	93.53	1.43	28.76
2007	1.50	22.39	99.09	1.83	14.78	291.97	0.67	30.64	144.98	1.28	14.95
2008	1.73	23.93	90.78	2.08	16.63	273.38	0.87	31.71	139.62	1.51	15.90
2009	1.54	15.03	42.80	1.84	12.50	174.15	0.66	23.54	74.39	1.30	10.93
2010	1.66	10.78	54.20	1.94	10.97	174.50	0.74	21.48	85.18	1.46	7.92
2011	1.55	21.35	42.40	1.90	22.10	127.21	0.59	50.05	55.72	1.30	16.73
2012	1.42	1.67	35.93	1.85	1.77	131.75	0.42	4.69	45.10	1.14	1.35
2013	1.29	13.64	72.46	1.71	14.40	280.54	0.32	47.37	89.76	1.00	11.36
2014	1.07	5.88	14.72	1.46	6.82	165.67	0.20	28.34	30.11	0.81	5.21
2015	0.87	26.68	59.47	1.19	32.14	381.35	0.15	154.95	85.47	0.66	26.10
2016	0.79	2.52	48.92	1.05	2.69	366.80	0.12	13.64	69.33	0.59	2.38
2017	0.76	15.44	183.76	1.02	20.61	1208.23	0.12	105.68	230.24	0.56	14.56
2018	0.64	15.28	64.17	0.86	24.62	749.41	0.08	129.34	115.37	0.48	15.15
2019	0.62	3.80	136.25	0.83	4.21	1151.10	0.09	26.01	181.87	0.47	3.31

ASM applied to whiting data

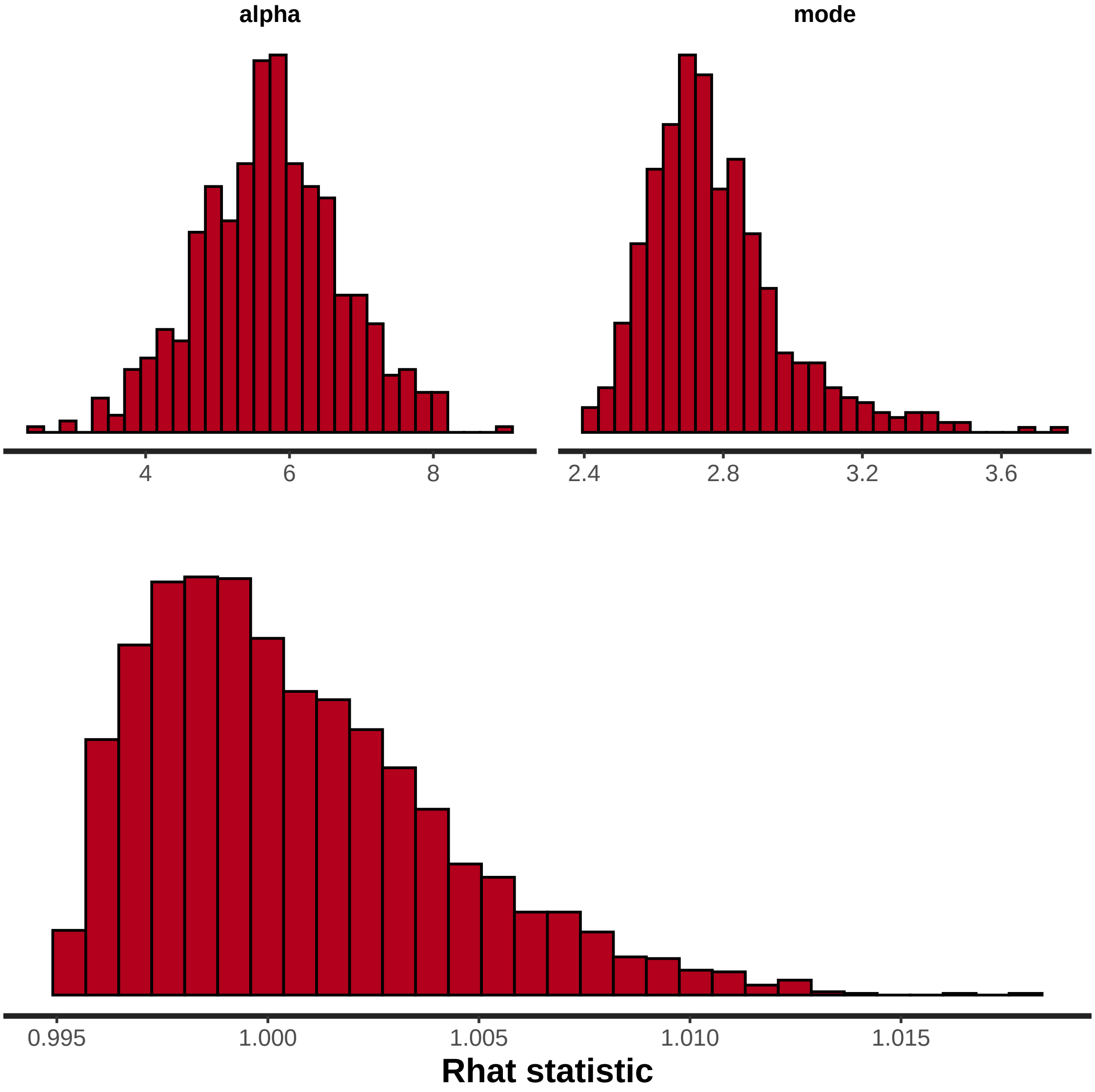
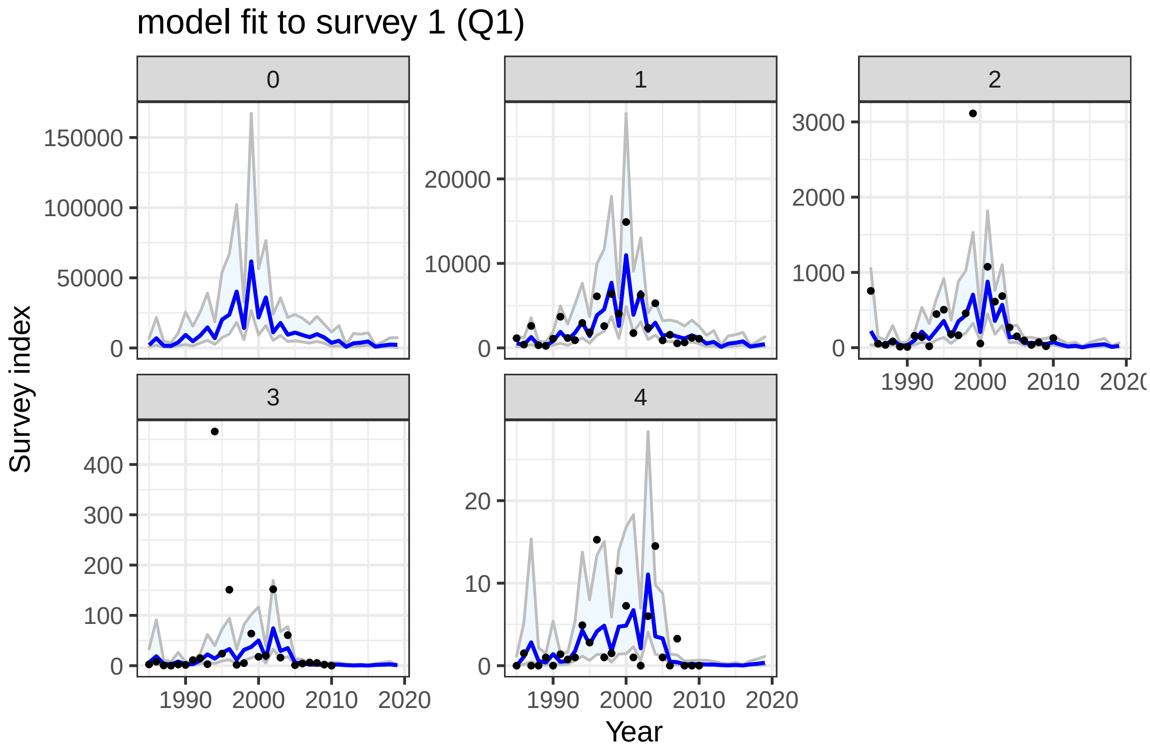


Figure D.7 - Histograms of critical parameters and Rhat value for the reference model applied to whiting Clyde data.


model fit to survey 1 (Q1)
0
1
2
3
4
150000
100000
50000
0
20000
10000
0
3000
2000
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0
400
300
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0
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10
0
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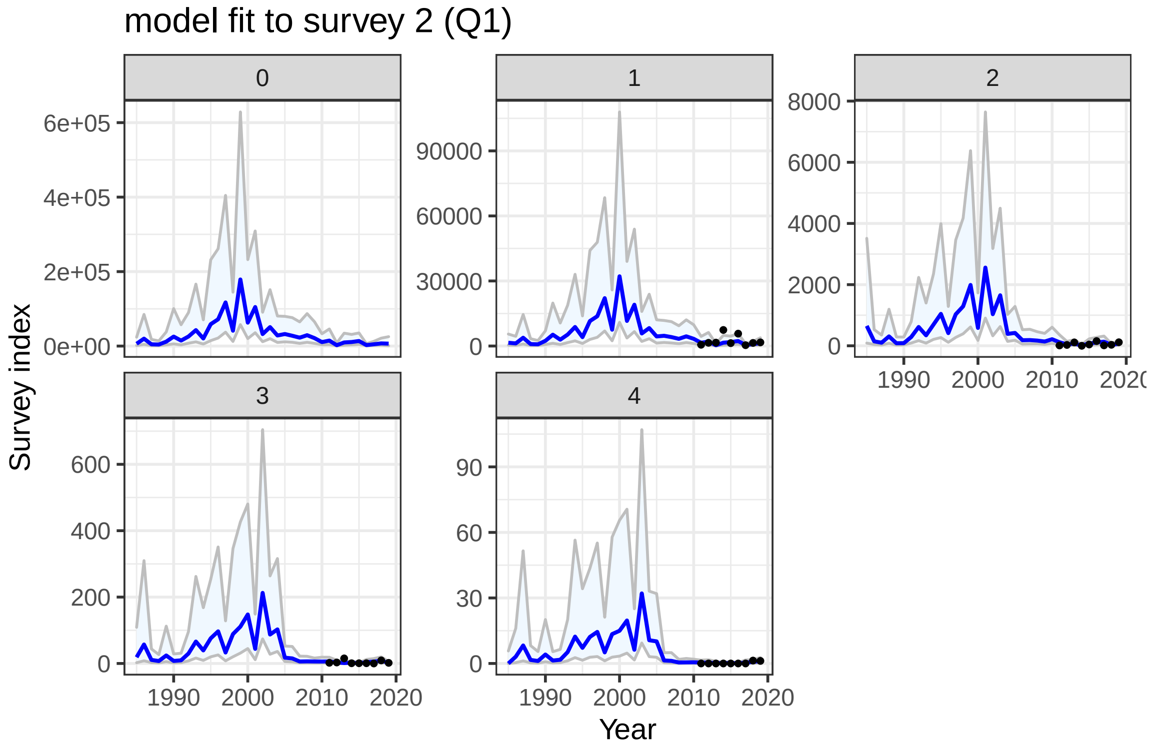
2000
2010
2020
Survey index
Year


model fit to survey 2 (Q1)
0
1
2
3
4
6e+05
4e+05
2e+05
0e+00
90000
60000
30000
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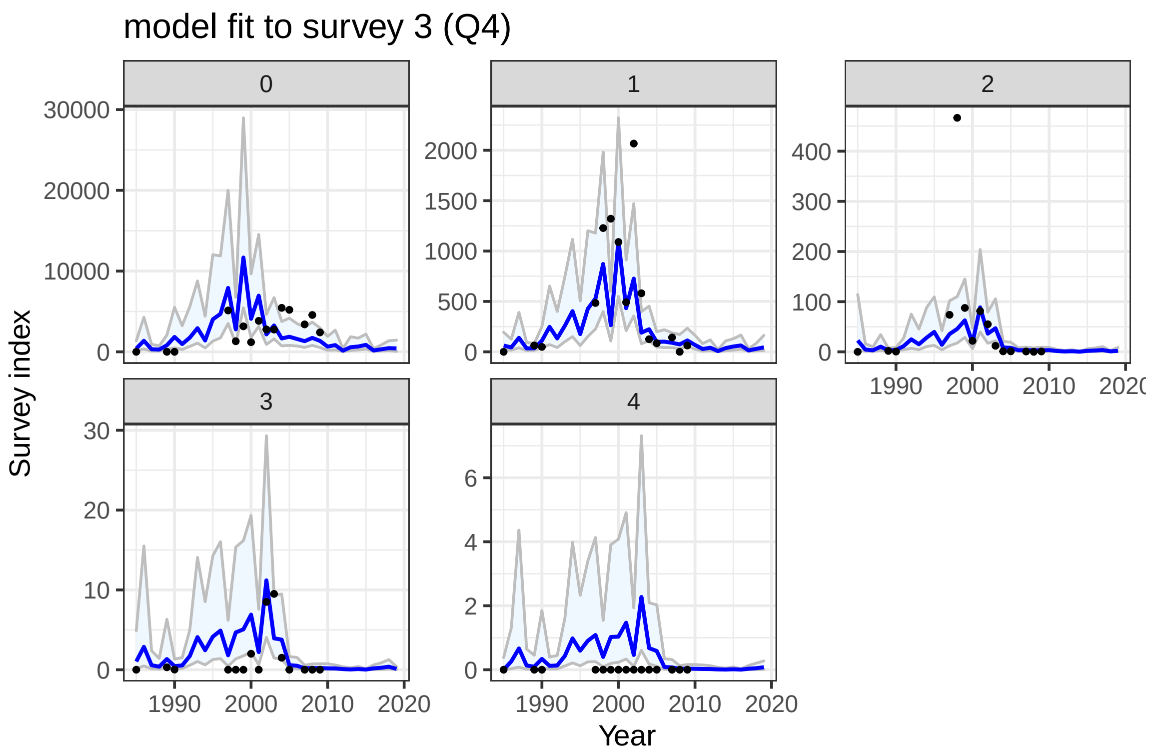
2000
2010
2020
Survey index
Year



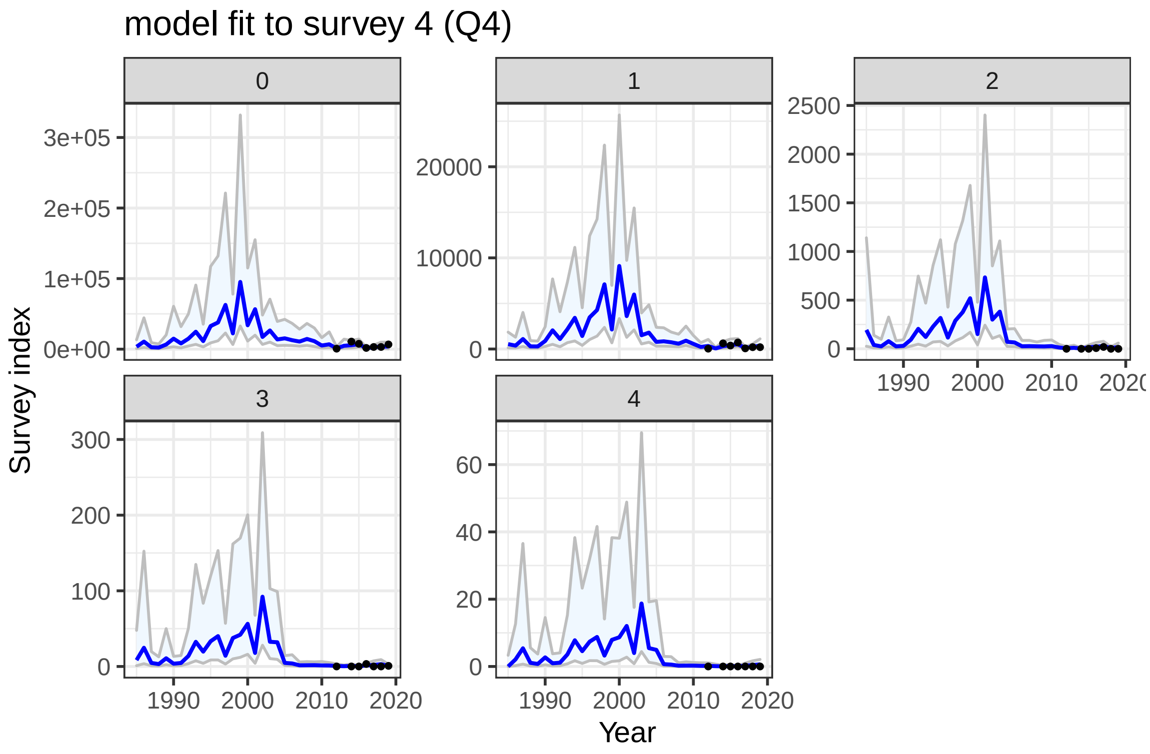


Figure D.8 - Model fit to survey data for the reference model applied to whiting Clyde data for ages 0-4.

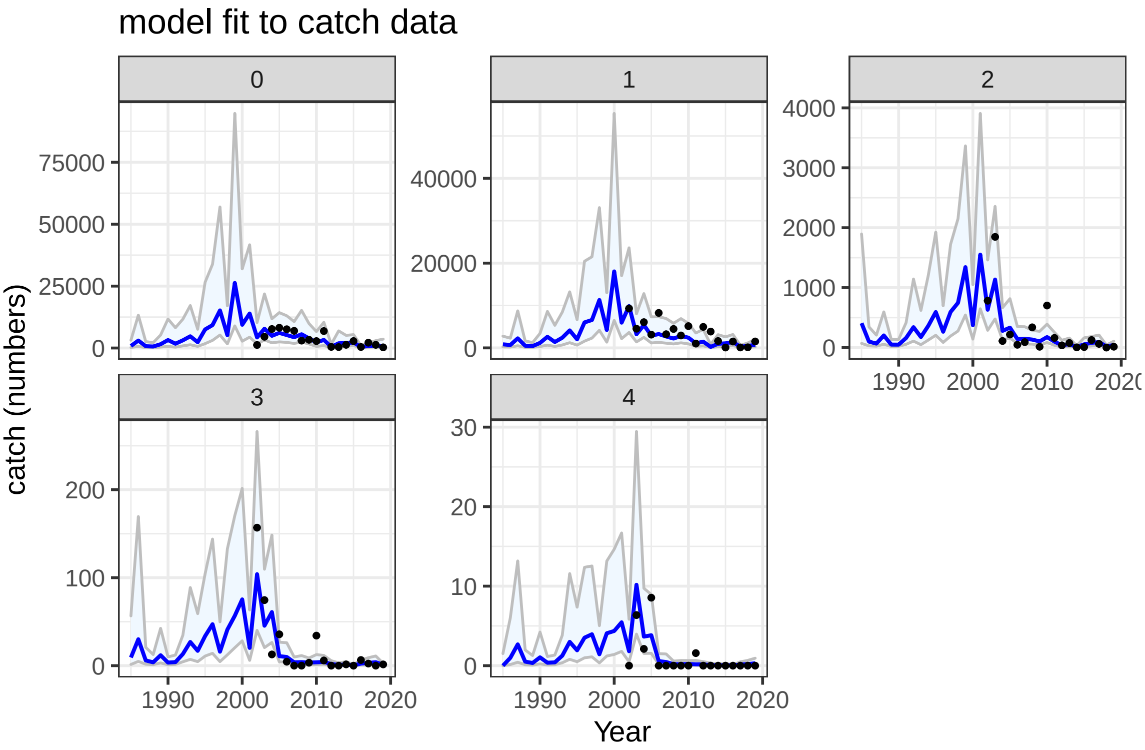


Figure D.9 - Model fit to catch data for the reference model applied to whiting Clyde data for ages 0-4.

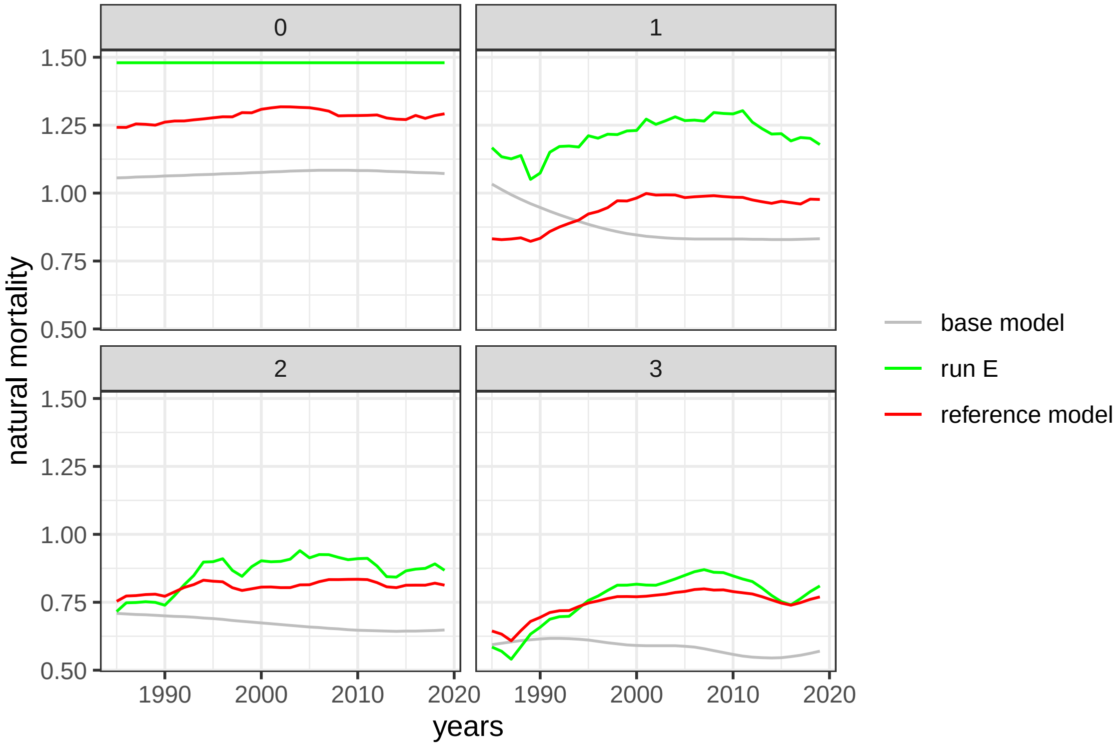


Figure D.10 - Comparison of natural mortality at age values used for the base model, model run E and reference model applied to whiting Clyde data.

Table D.2 - Stock summary statistics for each model run/configuration using whiting data. B represents spawning stock biomass (in tonnes), F is mean fishing mortality, and R is a measure of recruitment (number of fish at age 0 in thousands). Base stands for base model, run Z represents the reference model run.

years	base_B	base_F	base_R	runA_B	runA_F	runA_R	runB_B	runB_F	runB_R	runC_B
1985	145.51	1.51	2.17	NA	NA	NA	146.63	1.74	2.58	234.43
1986	45.41	1.73	12.27	NA	NA	NA	43.86	1.94	13.61	78.11
1987	11.24	1.92	1.98	NA	NA	NA	10.44	2.04	2.12	18.40
1988	34.03	1.88	2.01	NA	NA	NA	32.39	1.95	2.23	56.20
1989	9.10	1.71	6.23	NA	NA	NA	9.61	1.80	6.66	17.18
1990	7.92	1.54	11.55	NA	NA	NA	8.23	1.66	11.75	12.90
1991	24.89	1.48	6.00	NA	NA	NA	23.70	1.57	6.24	39.82
1992	43.20	1.43	10.38	NA	NA	NA	40.49	1.49	11.11	75.57
1993	26.29	1.24	24.24	NA	NA	NA	25.95	1.33	27.44	41.16
1994	40.90	1.31	8.28	NA	NA	NA	39.97	1.38	8.55	55.87
1995	89.87	1.41	20.55	NA	NA	NA	94.31	1.53	21.49	130.69
1996	40.06	1.60	26.53	NA	NA	NA	39.24	1.69	29.36	61.23
1997	55.39	1.49	46.64	NA	NA	NA	53.50	1.59	49.33	100.86
1998	93.90	1.40	12.85	NA	NA	NA	95.95	1.49	15.02	142.61
1999	164.22	1.53	48.12	NA	NA	NA	156.61	1.63	51.79	257.54
2000	54.57	1.43	17.12	NA	NA	NA	57.33	1.50	17.54	97.63
2001	147.95	1.33	38.41	NA	NA	NA	149.13	1.39	39.68	268.37
2002	81.89	1.43	15.71	142.64	1.42	14.82	83.56	1.52	16.53	123.67
2003	120.03	1.90	24.45	141.74	1.74	18.96	116.36	1.99	24.76	170.76
2004	35.88	2.44	21.37	34.03	2.05	23.80	35.42	2.49	21.72	50.82
2005	26.33	2.78	24.25	22.08	2.31	26.24	25.19	2.90	25.40	37.77
2006	13.91	2.93	19.46	17.22	2.45	19.49	13.08	3.05	20.10	17.80
2007	12.50	2.72	13.28	14.87	2.32	15.16	11.49	2.80	13.95	17.50
2008	12.44	2.67	17.95	12.99	2.37	21.76	12.02	2.78	19.59	16.61
2009	9.66	2.32	13.24	10.48	2.09	12.36	9.23	2.48	14.37	11.19
2010	17.41	2.53	9.01	19.46	2.46	9.20	17.12	2.74	9.77	22.15
2011	12.15	2.69	18.29	8.28	2.38	19.04	10.59	2.81	19.60	12.02
2012	7.50	2.55	2.32	6.52	2.28	2.16	7.14	2.77	2.53	5.21
2013	18.06	2.47	5.15	16.50	2.23	5.12	15.92	2.68	5.96	9.26
2014	3.29	2.09	6.32	2.96	1.85	6.83	3.05	2.28	6.47	2.00
2015	7.89	1.83	13.15	7.20	1.61	12.65	7.58	1.98	13.27	8.27
2016	13.43	1.98	2.18	14.13	1.80	2.41	12.58	2.12	2.33	9.38
2017	24.63	1.93	5.73	21.02	1.67	5.70	22.37	2.00	5.84	13.87
2018	6.18	1.62	9.22	6.27	1.46	9.43	5.92	1.74	9.58	3.40

2019	15.57	1.50	2.45	14.03	1.36	2.70	14.21	1.61	2.69	6.57

years	runC_F	runC_R	runD_B	runD_F	runD_R	runE_B	runE_F	runE_R	runZ_B	runZ_F	runZ_R
1985	1.48	3.60	91.73	1.54	3.30	182.31	1.48	4.27	151.50	1.50	5.40
1986	1.65	19.66	33.79	1.68	9.71	60.96	1.74	24.91	59.36	1.58	18.27
1987	1.75	3.41	16.74	1.68	2.22	14.35	1.92	3.63	29.17	1.62	4.44
1988	1.72	3.21	33.72	1.58	2.13	43.57	1.81	3.65	62.41	1.55	3.75
1989	1.58	11.01	13.40	1.42	5.25	12.05	1.61	11.76	23.84	1.39	10.50
1990	1.52	21.59	12.24	1.31	11.27	10.65	1.45	22.13	19.45	1.30	23.97
1991	1.45	9.72	26.31	1.30	7.02	32.78	1.37	10.83	48.87	1.27	12.79
1992	1.43	16.61	53.09	1.29	12.09	55.15	1.30	19.50	91.44	1.27	22.95
1993	1.29	38.85	35.35	1.17	20.48	32.16	1.08	48.40	56.00	1.17	38.19
1994	1.31	15.14	51.72	1.25	8.90	49.65	1.13	16.50	78.69	1.22	18.36
1995	1.38	36.03	83.13	1.40	26.25	115.40	1.22	37.92	118.54	1.32	53.20
1996	1.51	41.83	39.29	1.59	32.46	49.88	1.39	51.16	60.30	1.45	61.91
1997	1.46	84.48	75.55	1.52	51.55	74.35	1.29	91.25	120.74	1.40	104.94
1998	1.41	27.09	109.47	1.53	17.21	121.42	1.18	28.72	167.61	1.42	36.47
1999	1.51	101.62	155.84	1.73	77.87	203.54	1.29	103.00	246.85	1.58	166.80
2000	1.47	31.68	49.97	1.75	27.23	74.74	1.23	35.40	82.76	1.58	59.14
2001	1.43	65.88	157.84	1.53	46.71	195.00	1.11	72.52	268.27	1.45	94.44
2002	1.52	23.87	89.79	1.52	16.40	101.87	1.20	28.18	137.49	1.45	28.79
2003	1.84	33.92	129.99	1.86	27.68	144.62	1.63	41.34	186.57	1.68	47.05
2004	2.24	22.10	38.25	2.26	17.10	43.71	2.07	30.67	51.52	1.96	25.41
2005	2.45	26.39	35.89	2.45	21.16	31.47	2.41	35.13	48.06	2.15	29.23
2006	2.55	21.11	15.61	2.54	17.37	16.61	2.53	28.16	20.13	2.22	25.13
2007	2.40	14.12	16.00	2.55	14.37	14.81	2.33	19.74	19.83	2.19	20.08
2008	2.48	24.51	13.66	2.44	16.73	14.89	2.26	28.58	18.48	2.19	25.98
2009	2.31	17.14	12.32	2.18	12.76	11.26	2.02	21.49	15.31	2.02	20.16
2010	2.61	7.13	19.19	2.41	9.53	20.61	2.26	14.05	24.17	2.33	9.64
2011	2.65	11.17	13.33	2.67	17.84	13.77	2.35	28.45	13.97	2.49	13.74
2012	2.57	1.89	7.99	2.56	2.65	8.44	2.24	3.73	5.95	2.39	2.22
2013	2.50	7.95	16.85	2.52	6.09	21.40	2.18	8.86	9.97	2.28	9.03
2014	2.25	6.64	3.43	2.13	7.03	3.96	1.81	10.79	2.39	2.02	9.91
2015	2.09	10.90	8.61	1.87	12.60	10.03	1.57	21.76	9.31	1.82	12.05
2016	2.19	1.95	14.35	1.92	2.25	16.57	1.72	3.73	13.63	1.86	2.42
2017	2.07	3.63	24.36	1.78	5.59	29.06	1.57	9.64	16.25	1.74	4.53
2018	1.88	5.02	6.71	1.54	9.62	8.01	1.32	15.88	4.54	1.62	6.29
2019	1.85	4.60	16.18	1.46	2.47	18.76	1.23	4.42	8.20	1.58	5.95

ASM applied to cod data

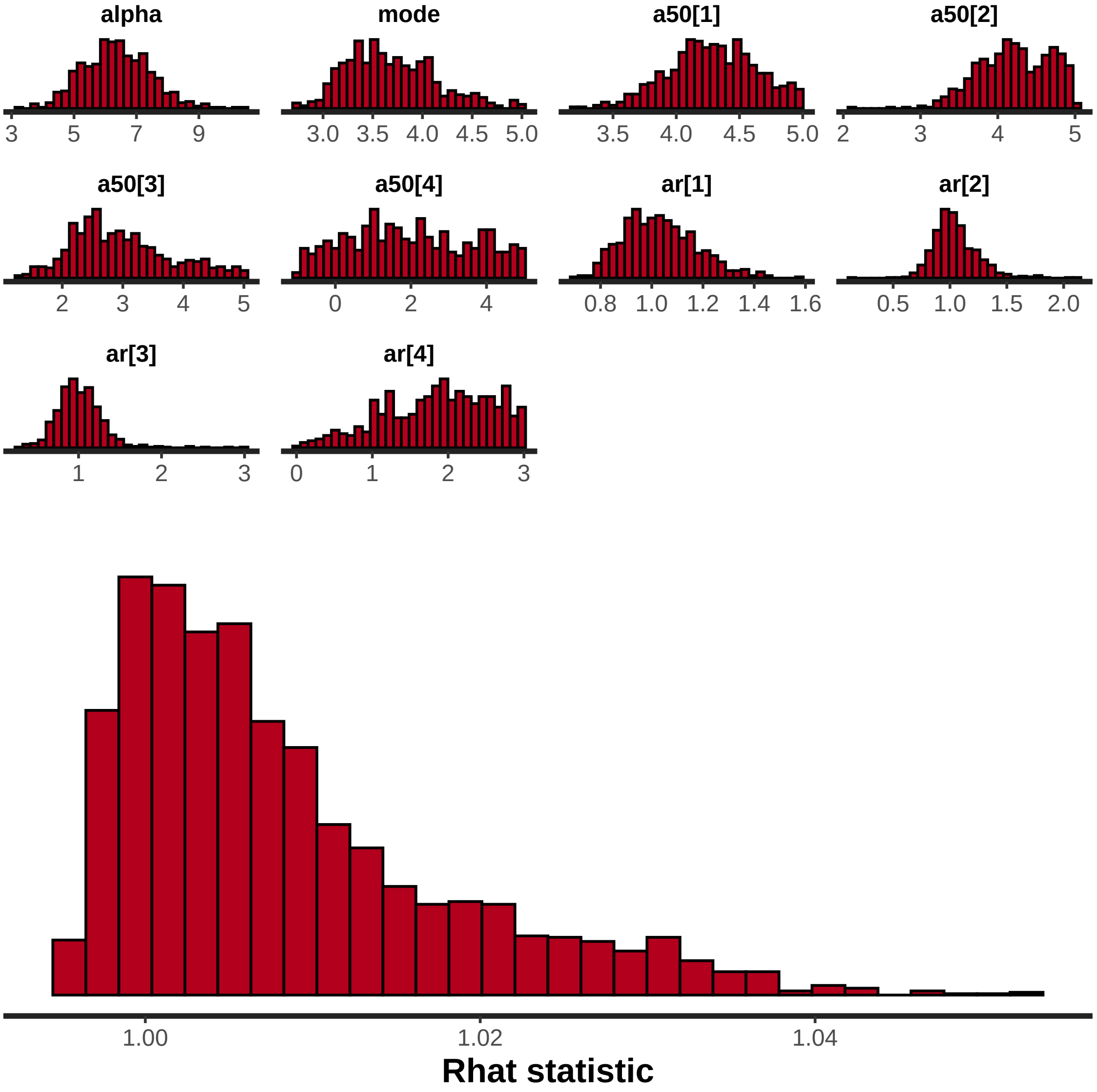
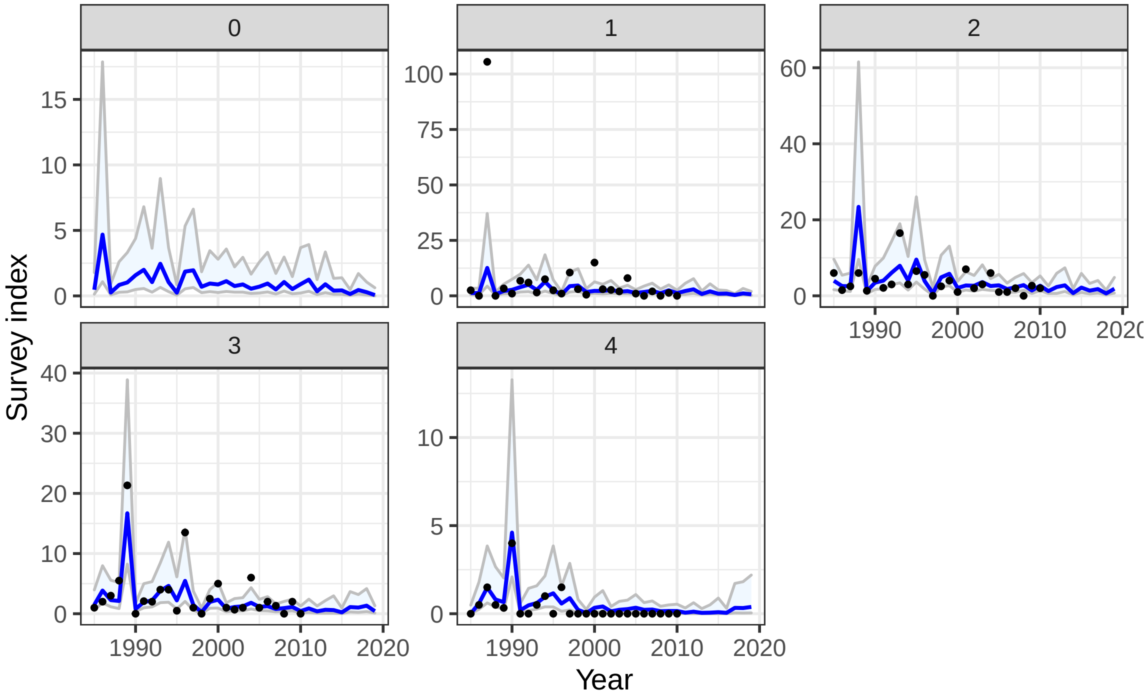
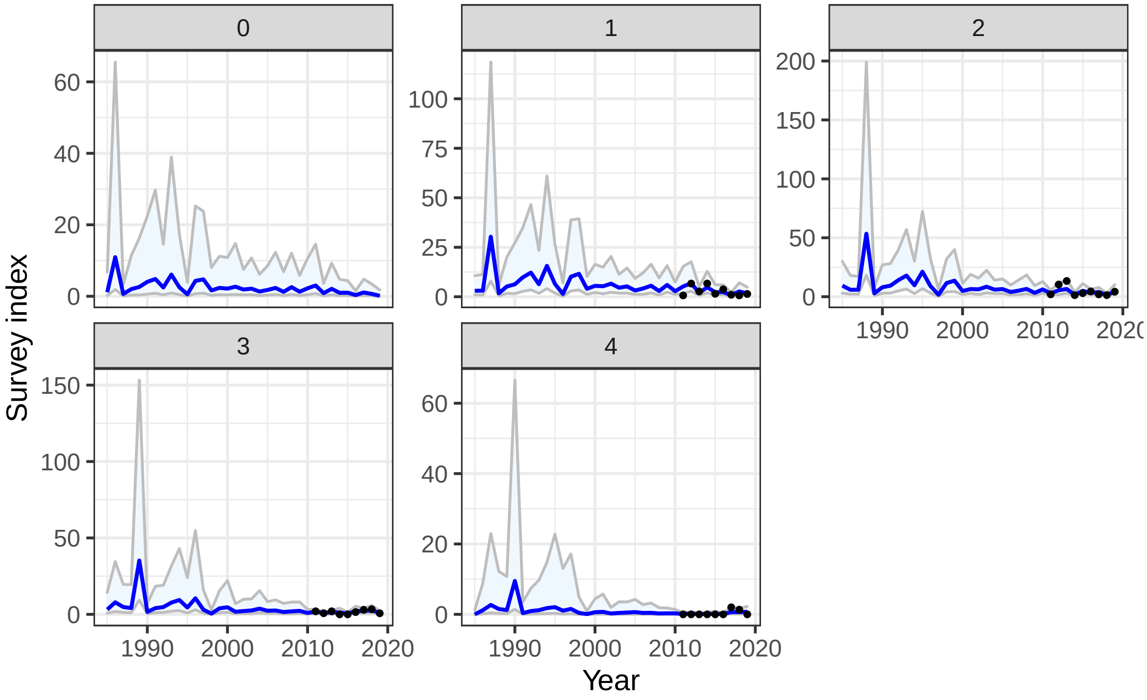


Figure D.11 - Histograms of critical parameters and Rhat value for the reference model applied to cod Clyde data.


model fit to survey 1 (Q1)


model fit to survey 2 (Q1)

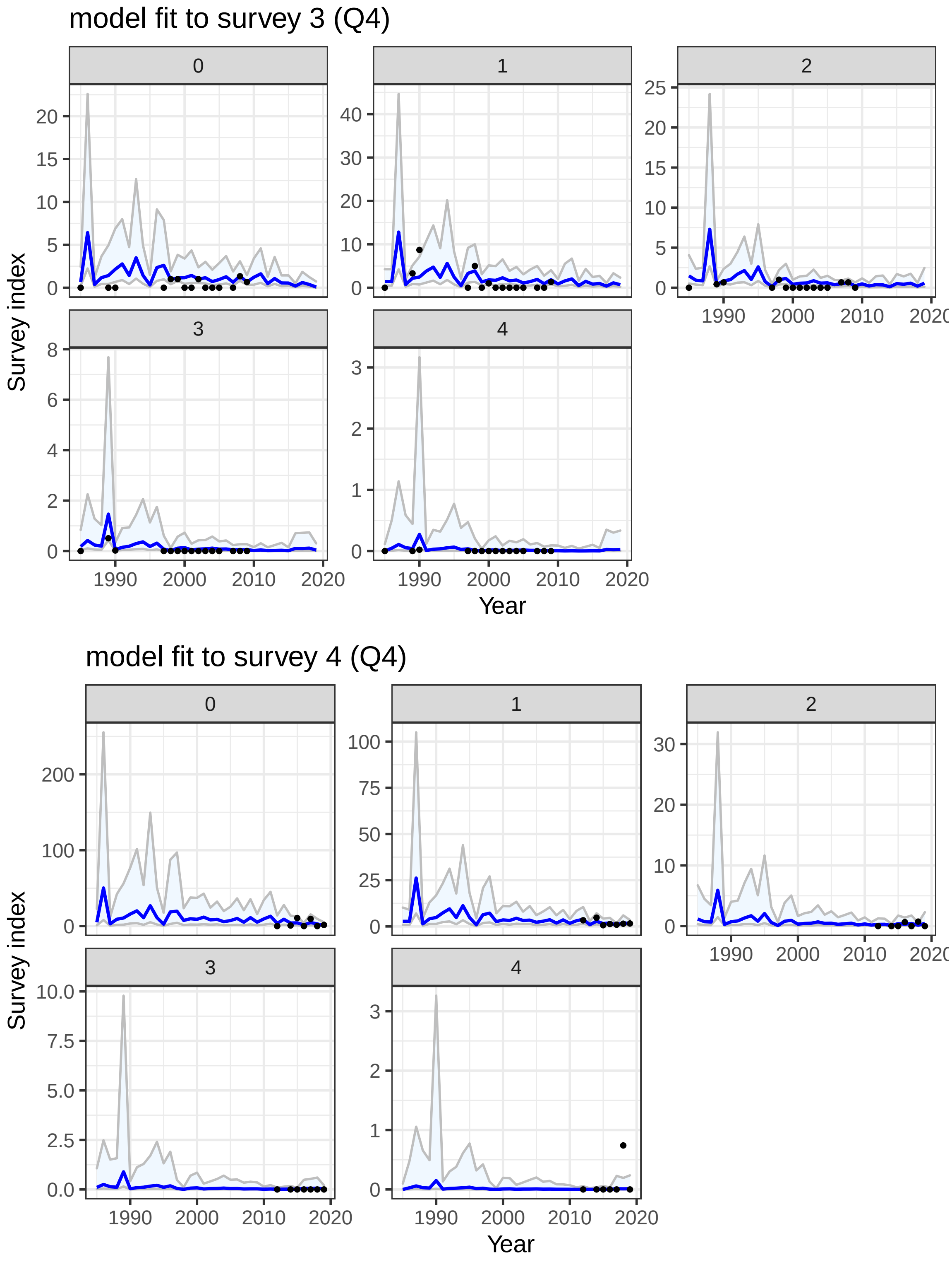


Figure D.12 - Model fit to survey data for the reference model applied to cod Clyde data for ages 0-4.

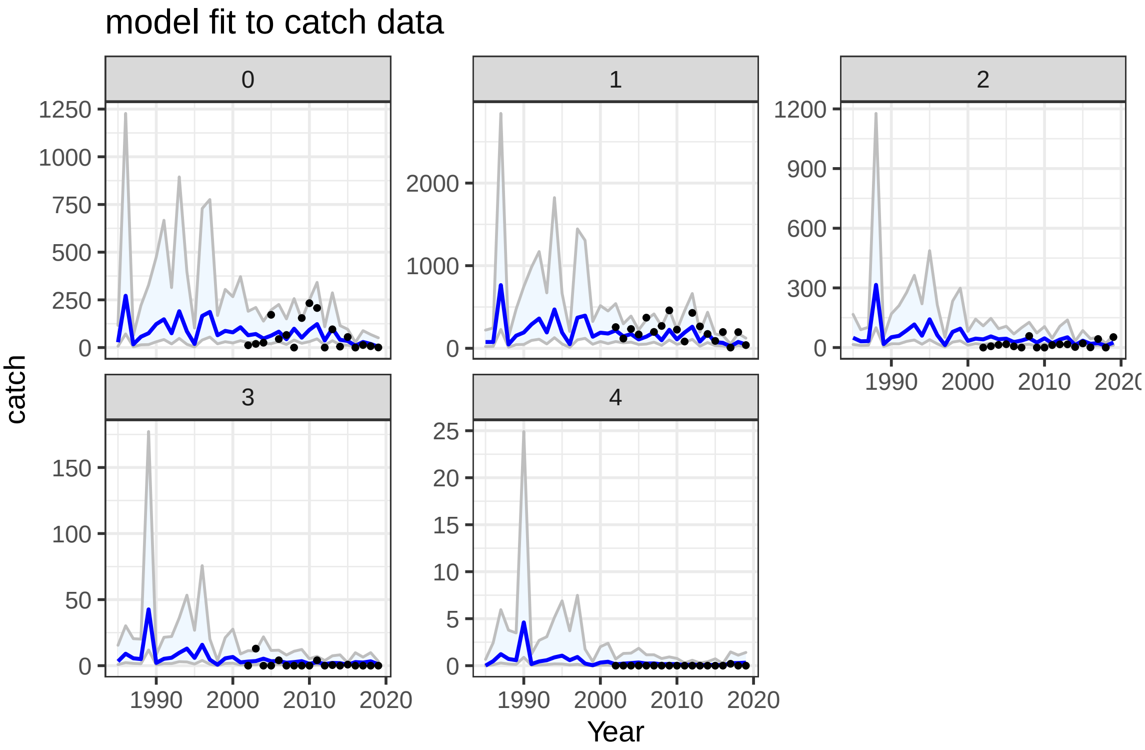


Figure D.13 - Model fit to catch data for the reference model applied to cod Clyde data for ages 0-4.

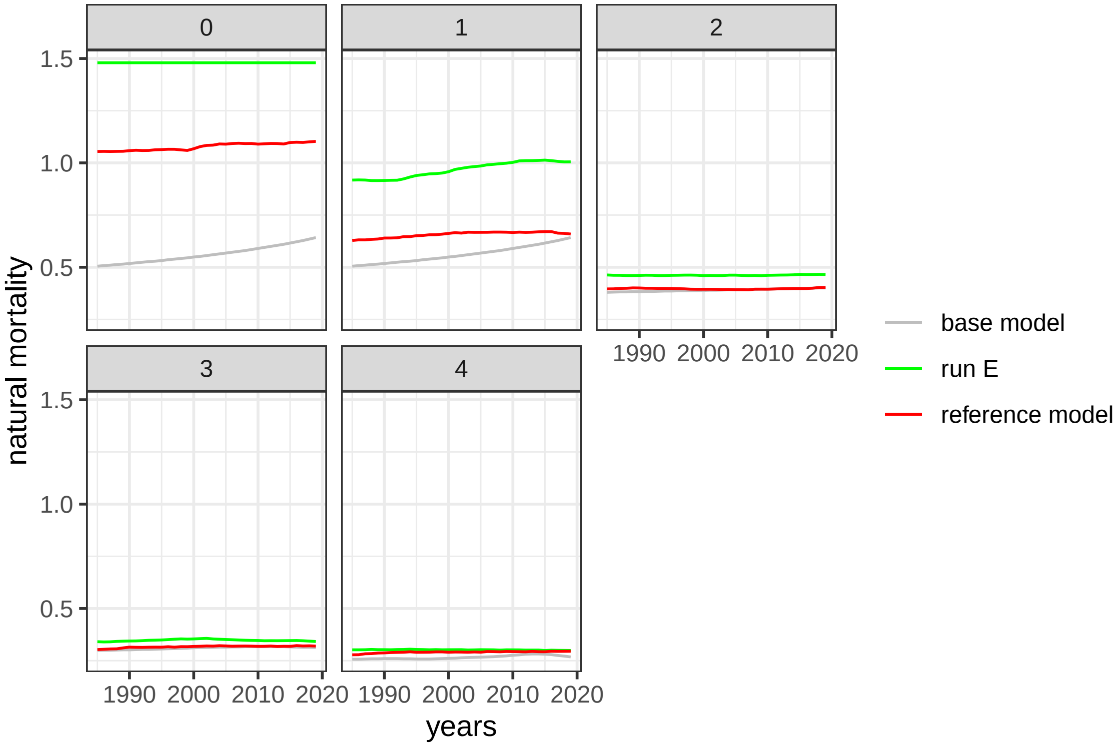


Figure D.14 - Comparison of natural mortality at age values used for the base model, model run E and reference model applied to cod Clyde data.

Table D.3 - Stock summary statistics for each model run/configuration using cod data. B represents spawning stock biomass (in tonnes), F is mean fishing mortality, and R is a measure of recruitment (number of fish at age 0 in thousands). Base stands for base model, run Z represents the reference model run.

years	base_B	base_F	base_R	runA_B	runA_F	runA_R	runB_B	runB_F	runB_R
1985	27.95	1.34	0.23	NA	NA	NA	22.34	1.47	0.26
1986	33.39	1.35	2.32	NA	NA	NA	24.87	1.46	2.16
1987	30.14	1.43	0.13	NA	NA	NA	22.93	1.54	0.14
1988	142.49	1.53	0.40	NA	NA	NA	97.30	1.58	0.38
1989	89.46	1.63	0.53	NA	NA	NA	65.76	1.67	0.48
1990	37.91	1.70	0.77	NA	NA	NA	28.68	1.74	0.78
1991	35.18	1.71	0.96	NA	NA	NA	23.35	1.74	1.01
1992	46.88	1.65	0.44	NA	NA	NA	33.08	1.73	0.46
1993	64.40	1.69	1.30	NA	NA	NA	45.96	1.74	1.26
1994	45.08	1.72	0.44	NA	NA	NA	34.55	1.78	0.46
1995	72.96	1.68	0.11	NA	NA	NA	48.50	1.79	0.11
1996	50.61	1.97	0.83	NA	NA	NA	34.25	2.01	0.84
1997	14.91	2.06	0.98	NA	NA	NA	11.27	2.06	0.95
1998	30.05	2.00	0.32	NA	NA	NA	18.98	1.97	0.35
1999	42.90	2.00	0.45	NA	NA	NA	28.70	2.01	0.48
2000	23.02	2.01	0.43	NA	NA	NA	17.13	2.06	0.43
2001	20.17	2.04	0.64	NA	NA	NA	14.37	2.06	0.61
2002	20.32	1.93	0.40	11.33	1.38	0.34	13.26	1.98	0.43
2003	28.43	1.87	0.50	35.39	1.53	0.58	17.88	1.93	0.56
2004	24.34	1.95	0.31	33.13	1.92	0.31	16.22	2.03	0.32
2005	23.56	1.94	0.39	27.30	1.78	0.33	16.15	2.01	0.47
2006	18.05	1.96	0.52	19.88	1.75	0.53	12.45	2.00	0.58
2007	18.34	1.95	0.24	18.64	1.84	0.24	13.34	2.02	0.30
2008	23.10	1.99	0.51	24.14	1.77	0.49	16.13	2.05	0.55
2009	15.61	2.04	0.30	16.96	1.91	0.27	11.43	2.09	0.34
2010	19.02	1.94	0.64	19.65	1.69	0.52	13.34	1.99	0.57
2011	16.41	1.95	0.91	18.54	1.76	1.07	11.71	1.96	0.88
2012	25.05	2.04	0.31	24.63	1.96	0.39	16.56	2.03	0.32
2013	34.89	2.17	0.88	40.85	2.36	1.04	23.08	2.06	0.76
2014	17.31	1.94	0.40	16.77	1.93	0.39	13.33	1.90	0.36
2015	29.16	1.73	0.45	32.47	1.70	0.45	19.83	1.72	0.48
2016	28.05	1.51	0.16	26.84	1.32	0.17	19.16	1.55	0.14
2017	30.43	1.43	0.50	34.36	1.28	0.47	22.97	1.46	0.51
2018	21.99	1.44	0.27	26.44	1.37	0.35	17.11	1.45	0.23

2019	28.89	1.42	0.07	30.06	1.31	0.10	22.79	1.43	0.10

years	runC_B	runC_F	runC_R	runE_B	runE_F	runE_R	runZ_B	runZ_F	runZ_R
1985	52.52	1.31	0.34	25.12	1.28	0.69	49.54	1.24	0.62
1986	59.67	1.31	3.62	29.92	1.30	6.80	60.80	1.22	5.96
1987	51.87	1.39	0.19	27.73	1.38	0.40	51.92	1.30	0.34
1988	249.25	1.48	0.64	137.00	1.47	1.24	233.85	1.41	1.06
1989	159.56	1.60	0.75	81.21	1.59	1.63	157.67	1.51	1.29
1990	66.84	1.69	1.21	34.72	1.72	2.19	68.69	1.62	1.97
1991	58.20	1.68	1.60	34.42	1.70	2.84	56.41	1.60	2.53
1992	79.70	1.63	0.82	45.14	1.62	1.24	79.70	1.54	1.34
1993	120.61	1.67	2.03	63.90	1.67	3.67	112.93	1.60	3.24
1994	91.53	1.70	0.81	42.98	1.71	1.31	85.83	1.64	1.35
1995	125.85	1.66	0.20	67.06	1.63	0.32	122.55	1.59	0.31
1996	94.31	2.02	1.35	47.65	1.94	2.36	88.40	1.95	2.28
1997	26.81	2.15	1.54	13.26	2.05	2.74	25.82	2.04	2.53
1998	52.12	2.02	0.55	28.17	1.94	0.97	50.42	1.93	0.90
1999	74.15	1.99	0.78	43.04	1.95	1.29	72.53	1.93	1.19
2000	40.80	2.02	0.72	22.55	1.99	1.16	41.12	1.91	1.14
2001	38.80	2.06	0.82	19.80	2.03	1.89	36.89	1.94	1.48
2002	35.33	1.87	0.61	18.60	1.89	1.16	34.70	1.78	1.00
2003	44.20	1.79	0.70	29.47	1.83	1.49	45.03	1.71	1.12
2004	40.60	1.91	0.44	24.46	1.97	0.86	42.08	1.85	0.72
2005	38.96	1.92	0.61	22.94	1.93	1.25	38.61	1.83	0.92
2006	28.40	1.92	0.77	16.96	1.94	1.69	29.28	1.82	1.25
2007	30.65	1.99	0.41	19.40	1.96	0.76	30.22	1.87	0.64
2008	36.52	2.05	0.86	24.31	2.01	1.57	36.87	1.93	1.39
2009	24.65	2.25	0.41	15.01	2.13	0.90	25.87	2.11	0.67
2010	30.02	2.23	0.69	18.63	1.99	1.80	32.24	2.11	1.20
2011	19.32	2.20	0.96	15.62	1.97	2.69	20.69	2.14	1.58
2012	23.88	2.26	0.29	25.43	2.09	0.85	26.49	2.20	0.47
2013	33.63	2.51	0.69	34.47	2.22	2.28	34.62	2.40	1.13
2014	12.98	2.18	0.30	15.95	1.98	1.07	13.35	2.12	0.53
2015	21.54	1.82	0.32	28.85	1.73	1.22	22.70	1.80	0.53
2016	19.74	1.44	0.11	24.63	1.49	0.41	21.75	1.42	0.18
2017	23.99	1.39	0.36	27.78	1.42	1.39	24.93	1.37	0.57
2018	17.30	1.52	0.22	19.40	1.43	0.69	18.22	1.48	0.36
2019	21.60	1.50	0.04	28.08	1.37	0.20	22.62	1.45	0.08

Appendix E - Chapter 6

SDDM fit to test data

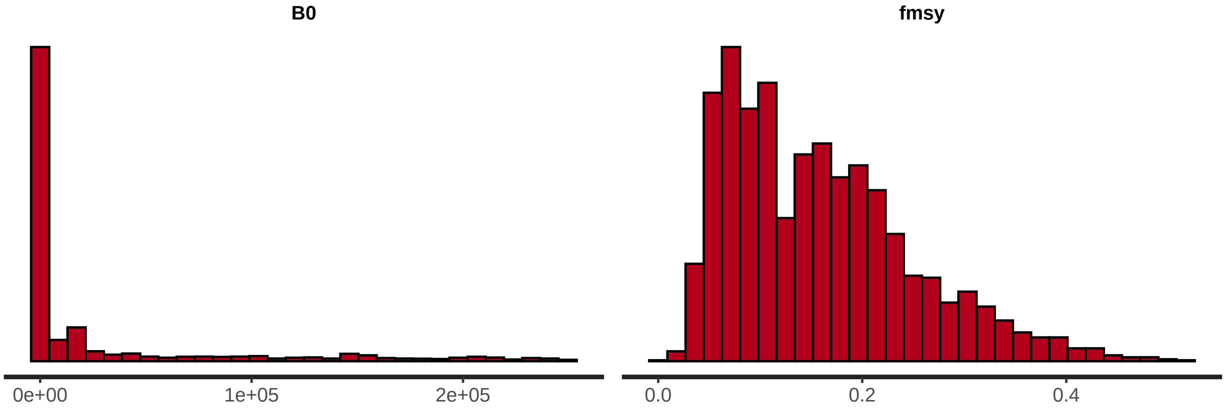


Figure E.1 - Histograms of B_0 and F_{msy} for the SDDM applied to test data.

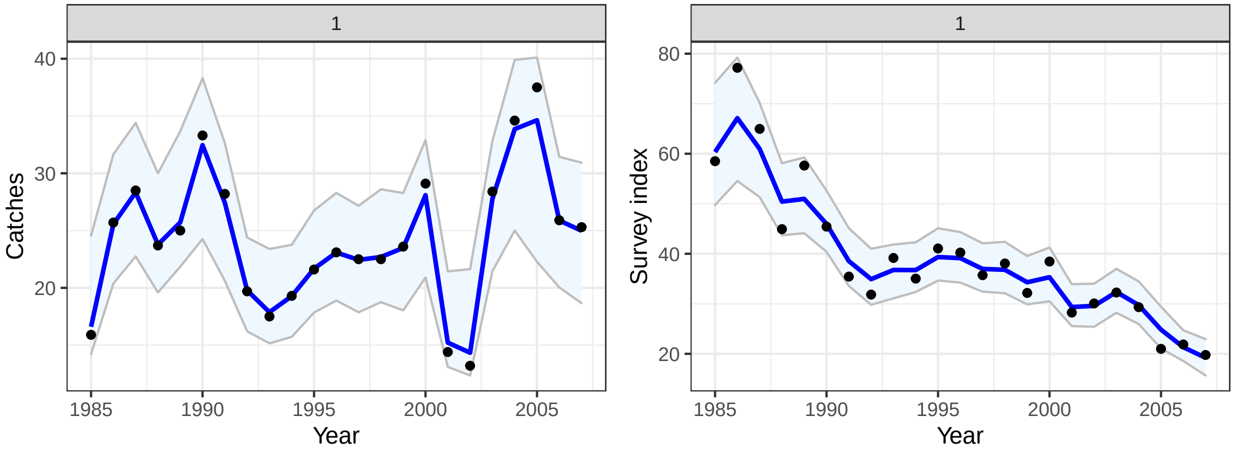


Figure E.2 - SDDM model fit to survey and catch test data. Blue line shows the model fit, shaded area the 0.95 credible interval and the dots are the data points.

Haddock data

Run 1: survey-only model

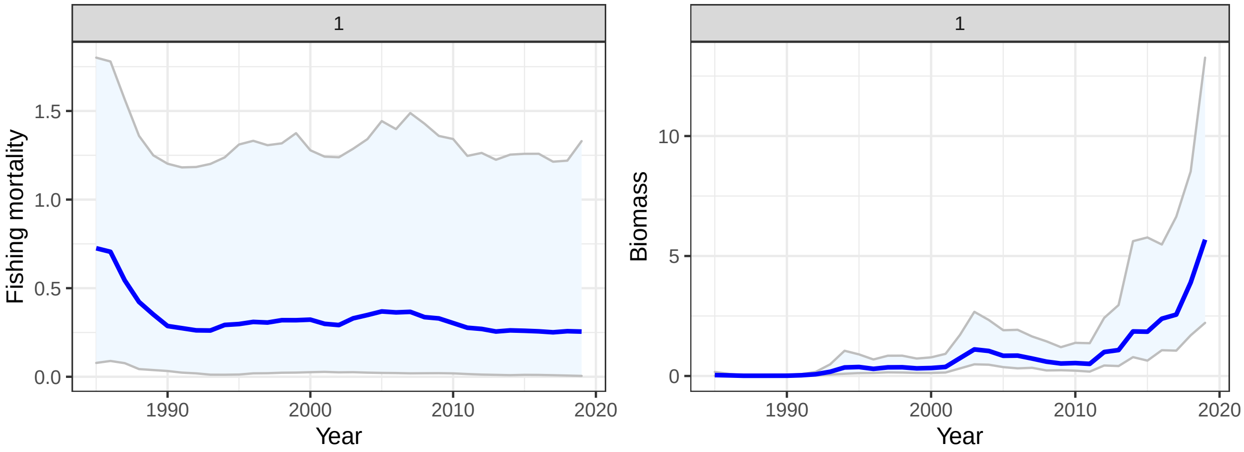


Figure E.3 - Estimated fishing mortality and stock biomass (log scale) for run 1 of the SDDM applied to haddock data. Shaded area represents 0.95 credible interval.

Run 2: catch data between 1990-2019

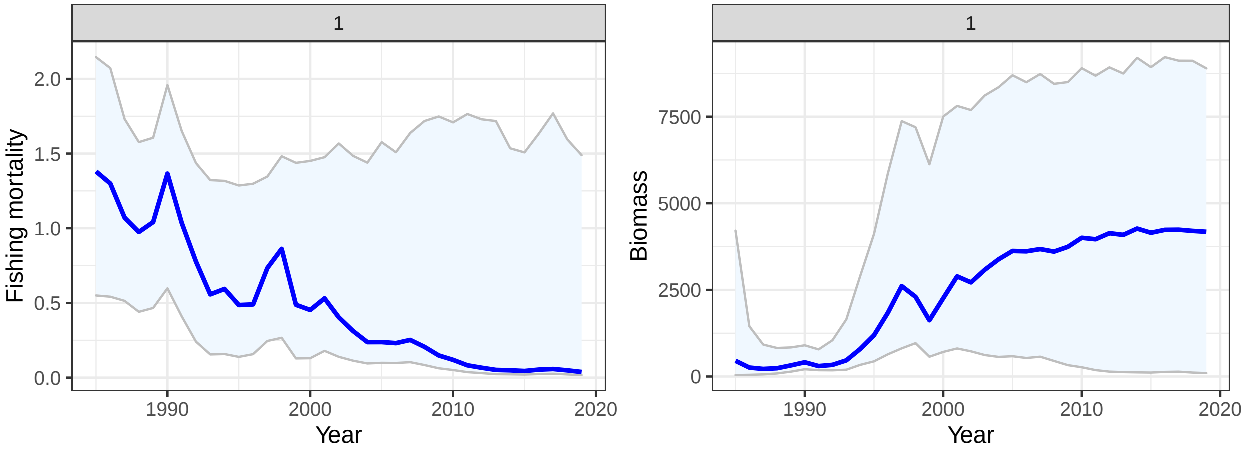


Figure E.4 - Estimated fishing mortality and stock biomass (in tonnes) for run 2 of the SDDM applied to haddock data. Shaded area represents 0.95 credible interval.

Run 4: fixed catch data between 2002-2019

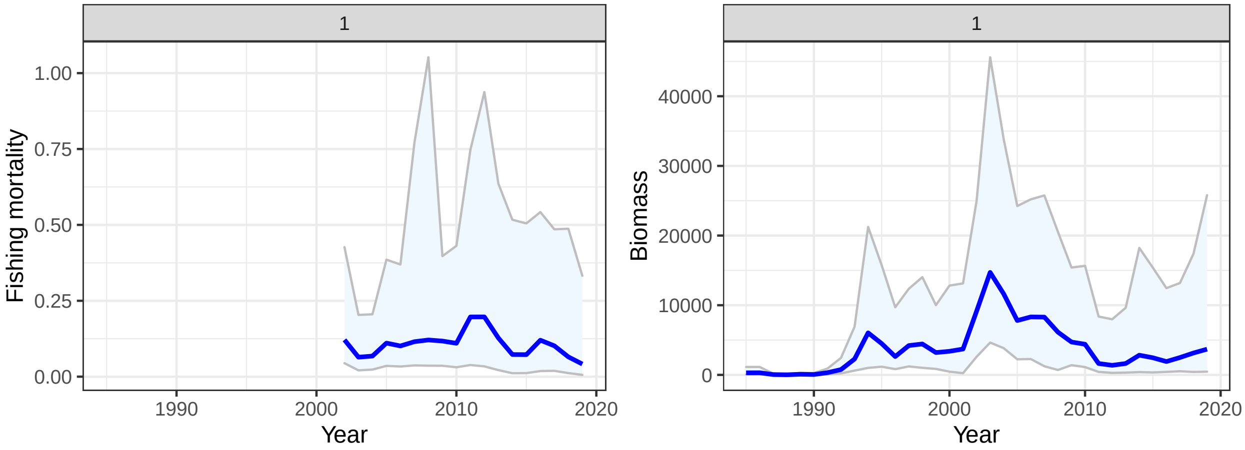


Figure E.5 - Estimated fishing mortality and stock biomass (in tonnes) for run 4 of the SDDM applied to haddock data. Shaded area represents 0.95 credible interval.

Whiting data

Run 1: survey-only model

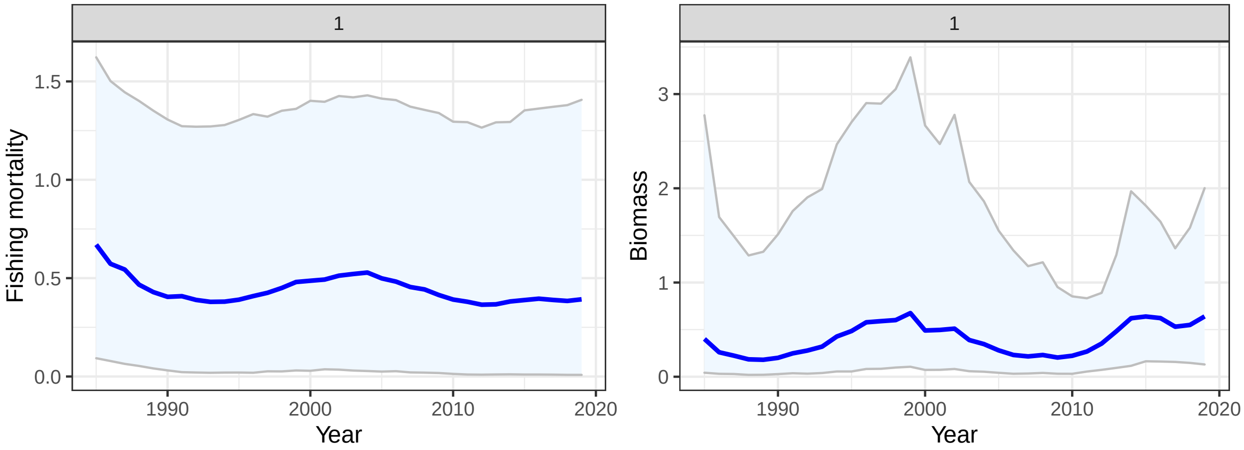


Figure E.6 - Estimated fishing mortality and stock biomass (log scale) for run 1 of the SDDM applied to whiting data. Shaded area represents 0.95 credible interval.

Run 2: catch data between 1990-2019

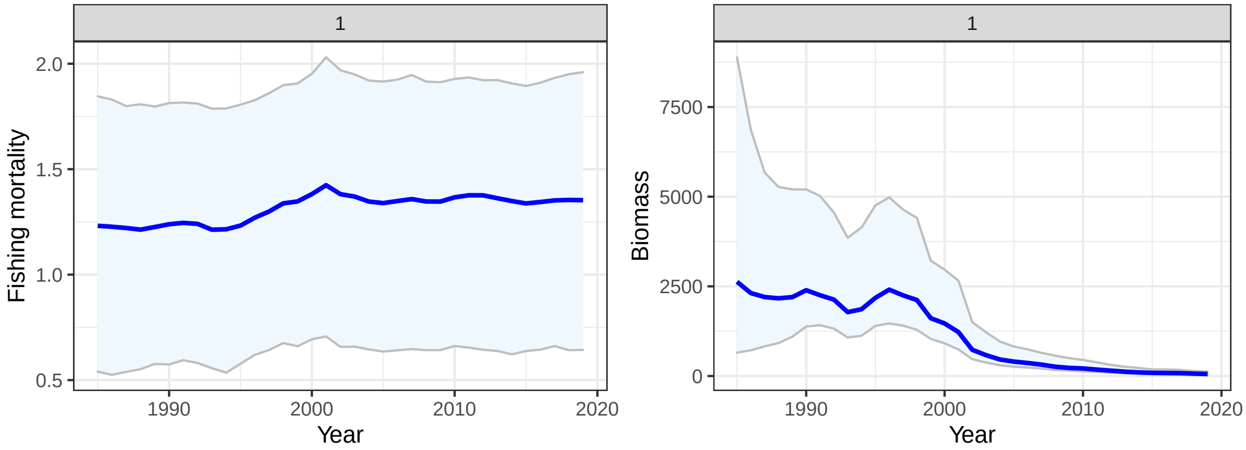


Figure E.7 - Estimated fishing mortality and stock biomass (in tonnes) for run 2 of the SDDM applied to whiting data. Shaded area represents 0.95 credible interval.

Run 4: fixed catch data between 2002-2019

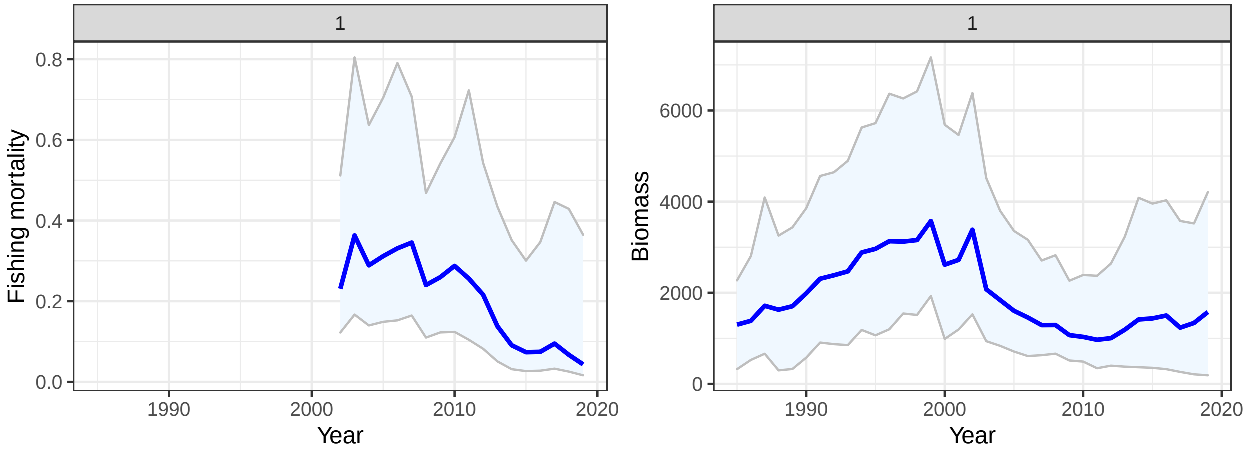


Figure E.8 - Estimated fishing mortality and stock biomass (in tonnes) for run 4 of the SDDM applied to whiting data. Shaded area represents 0.95 credible interval.

Cod data

Run 1: survey-only model

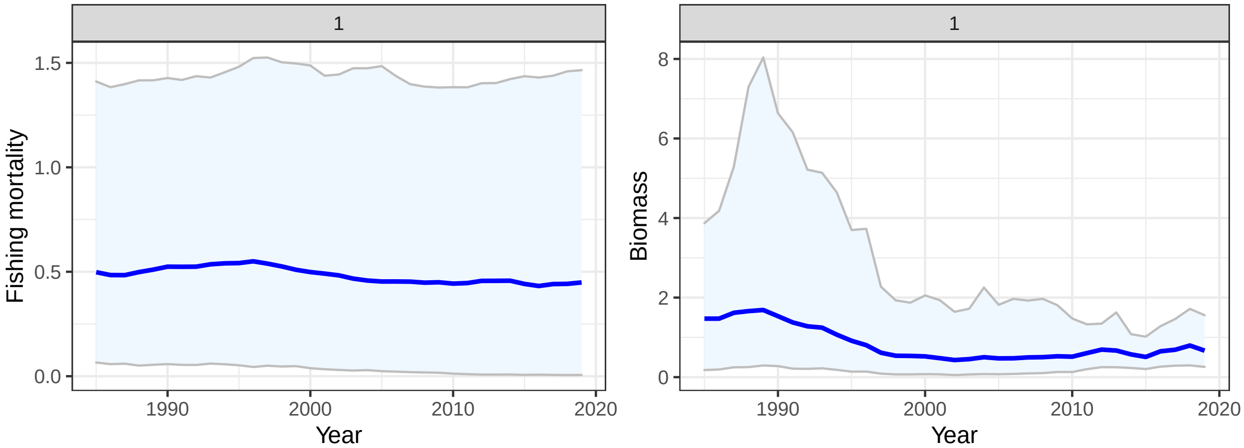


Figure E.9 - Estimated fishing mortality and stock biomass (log scale) for run 1 of the SDDM applied to cod data. Shaded area represents 0.95 credible interval.

Run 2: catch data between 1990-2019

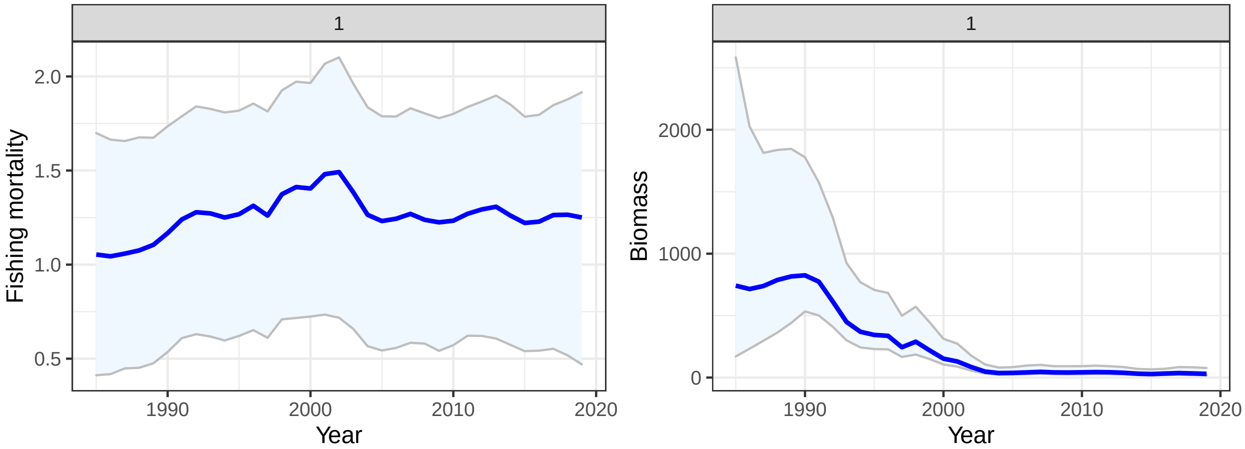


Figure E.10 - Estimated fishing mortality and stock biomass (in tonnes) for run 2 of the SDDM applied to cod data. Shaded area represents 0.95 credible interval.

Run 4: fixed catch data between 2002-2019

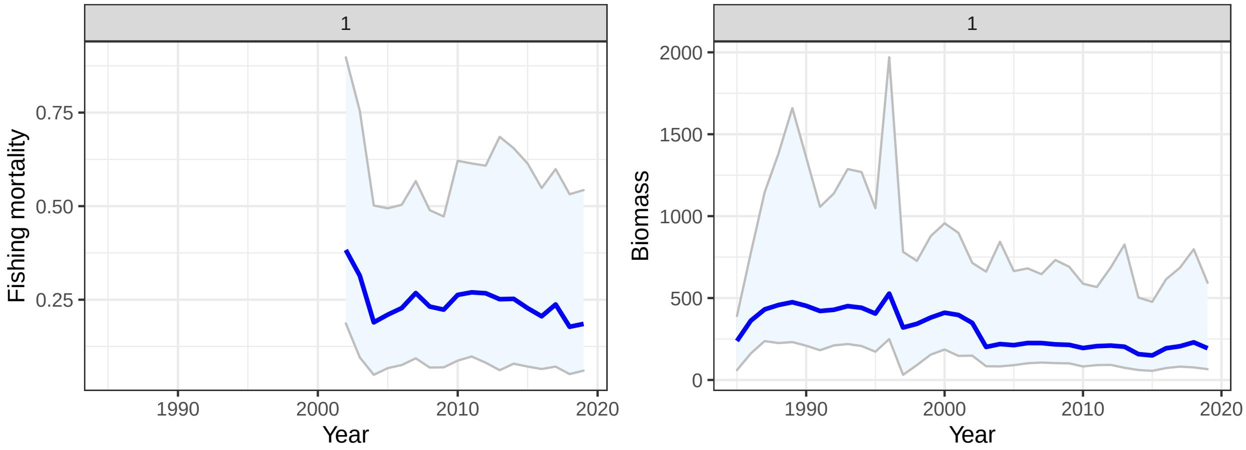


Figure E.11 - Estimated fishing mortality and stock biomass (in tonnes) for run 4 of the SDDM applied to cod data. Shaded area represents 0.95 credible interval.

Appendix F - Chapter 7

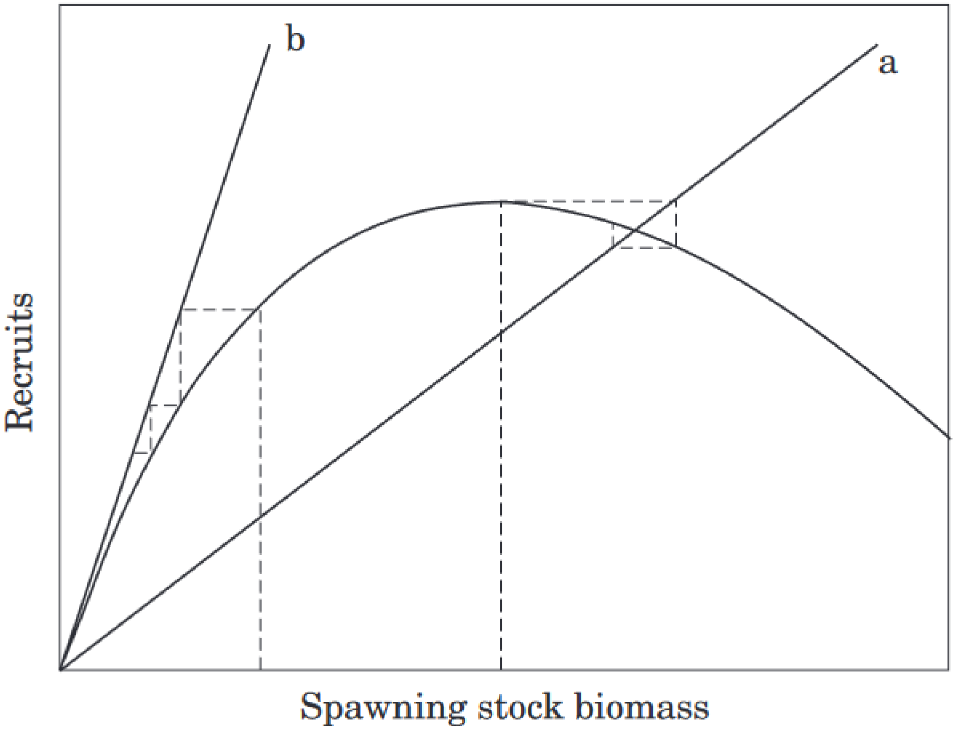


Figure 1. Theoretical stock–recruitment relationship for a fish stock. The curve predicts recruitment given stock size, while the straight lines are the replacement lines. These lines predict expected spawning stock from recruitment for two levels of exploitation. At a low exploitation rate (a), the dotted line indicates the population trajectory expected which cycles towards an equilibrium point. For high exploitation (b) the dotted line shows the expected population trajectory collapsing towards the origin.

Figure F.1 - Theoretical stock-recruitment relationship for a fish stock (from Cook, 1998).

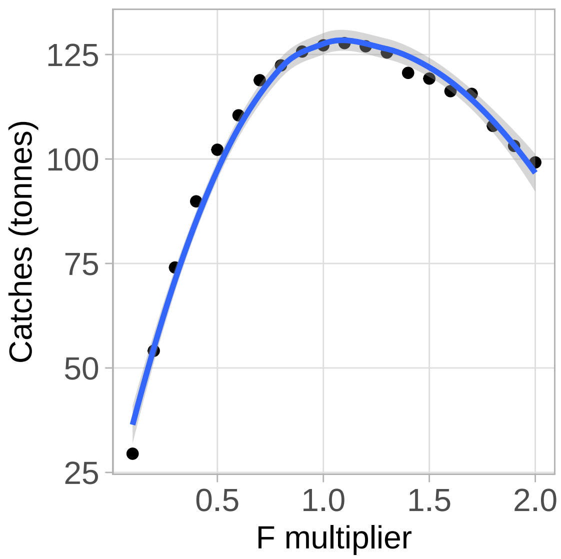


Figure F.2 - Median catch for the last 10 years of the projections (2038 until 2048) vs F multiplier for haddock data. Peak in the curve corresponds to MSY and F multiplier that gives Fmsy.

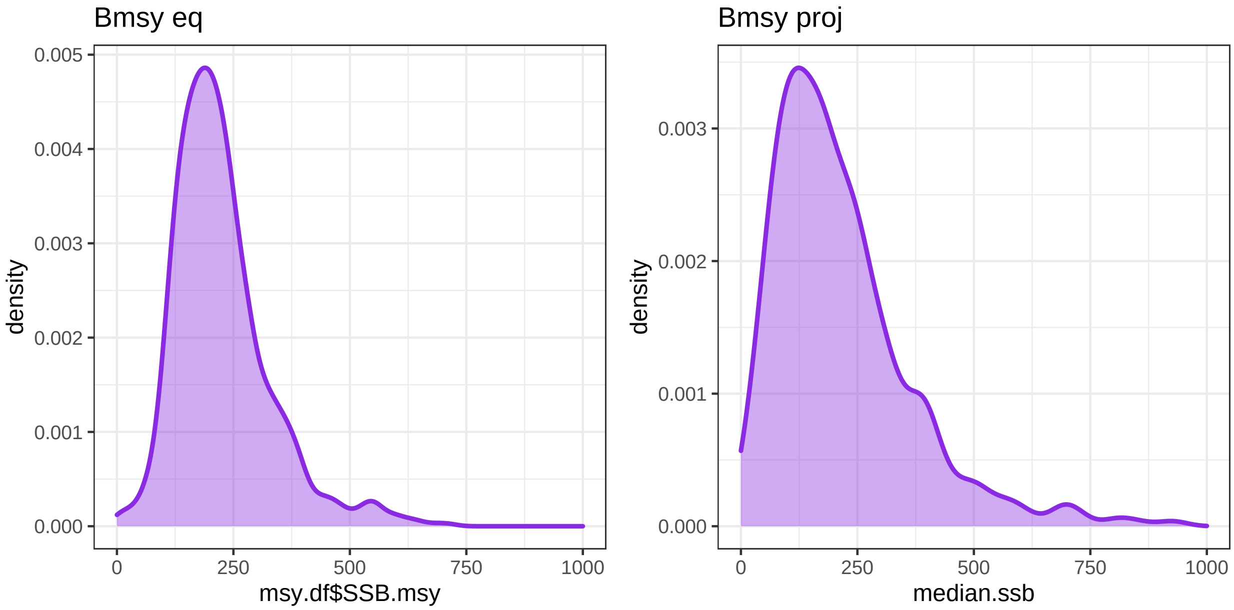


Figure F.3 - Biomass at MSY distributions estimated by equilibrium analysis (Bmsy eq) and by projection at Fmsy (Bmsy proj) for haddock data.

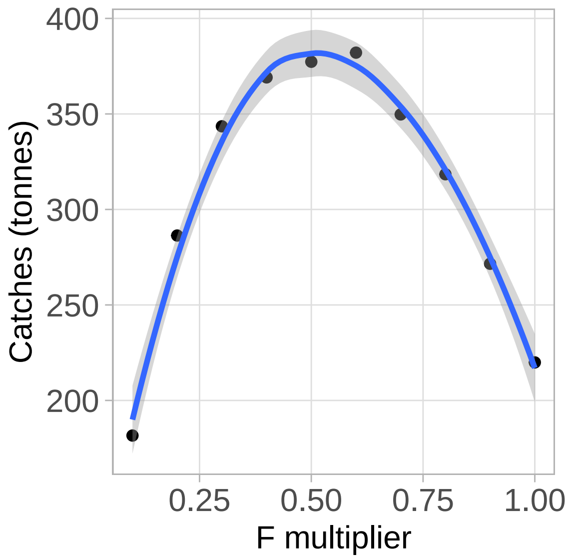


Figure F.4 - Median catch for the last 10 years of the projections (2038 until 2048) vs F multiplier for whiting data. Peak in the curve corresponds to MSY and F multiplier that gives Fmsy.

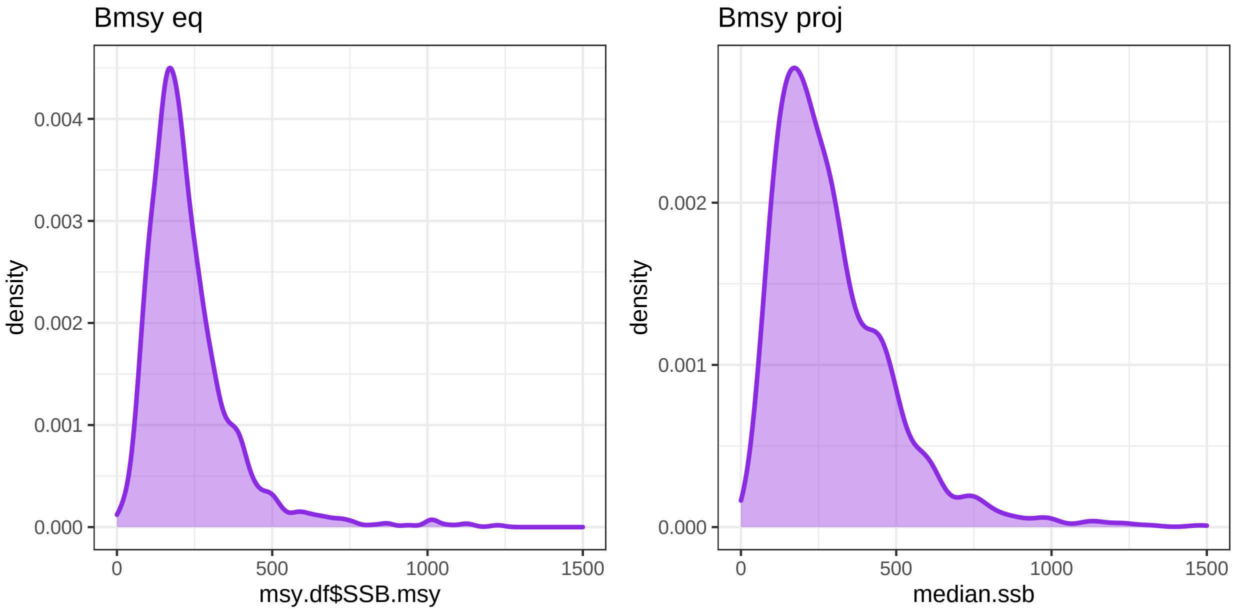


Figure F.5 - Biomass at MSY distributions estimated by equilibrium analysis (Bmsy eq) and by projection at Fmsy (Bmsy proj) for whiting data.

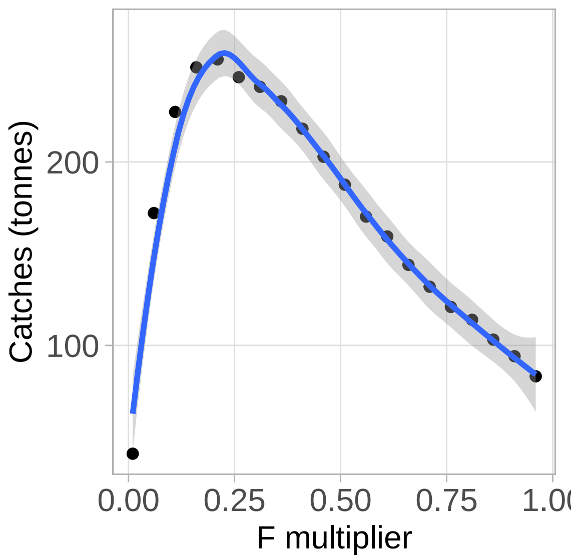


Figure F.6 - Median catch for the last 10 years of the projections (2038 until 2048) vs F multiplier for cod data. Peak in the curve corresponds to MSY and F multiplier that gives Fmsy.

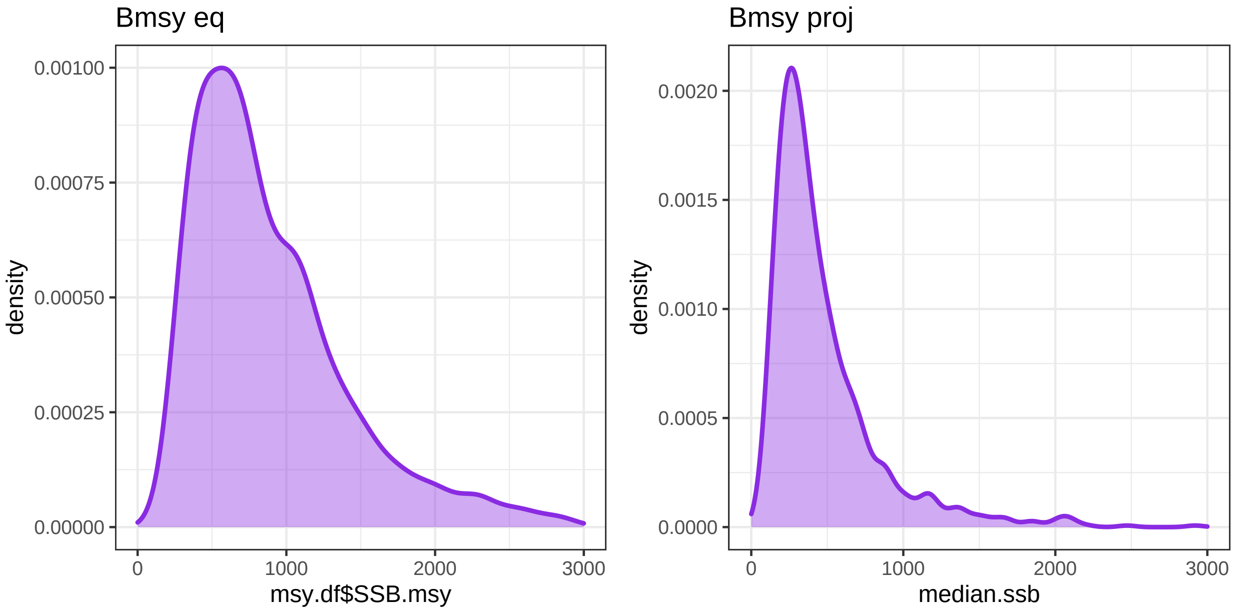


Figure F.7 - Biomass at MSY distributions estimated by equilibrium analysis (Bmsy eq) and by projection at Fmsy (Bmsy proj) for cod data.

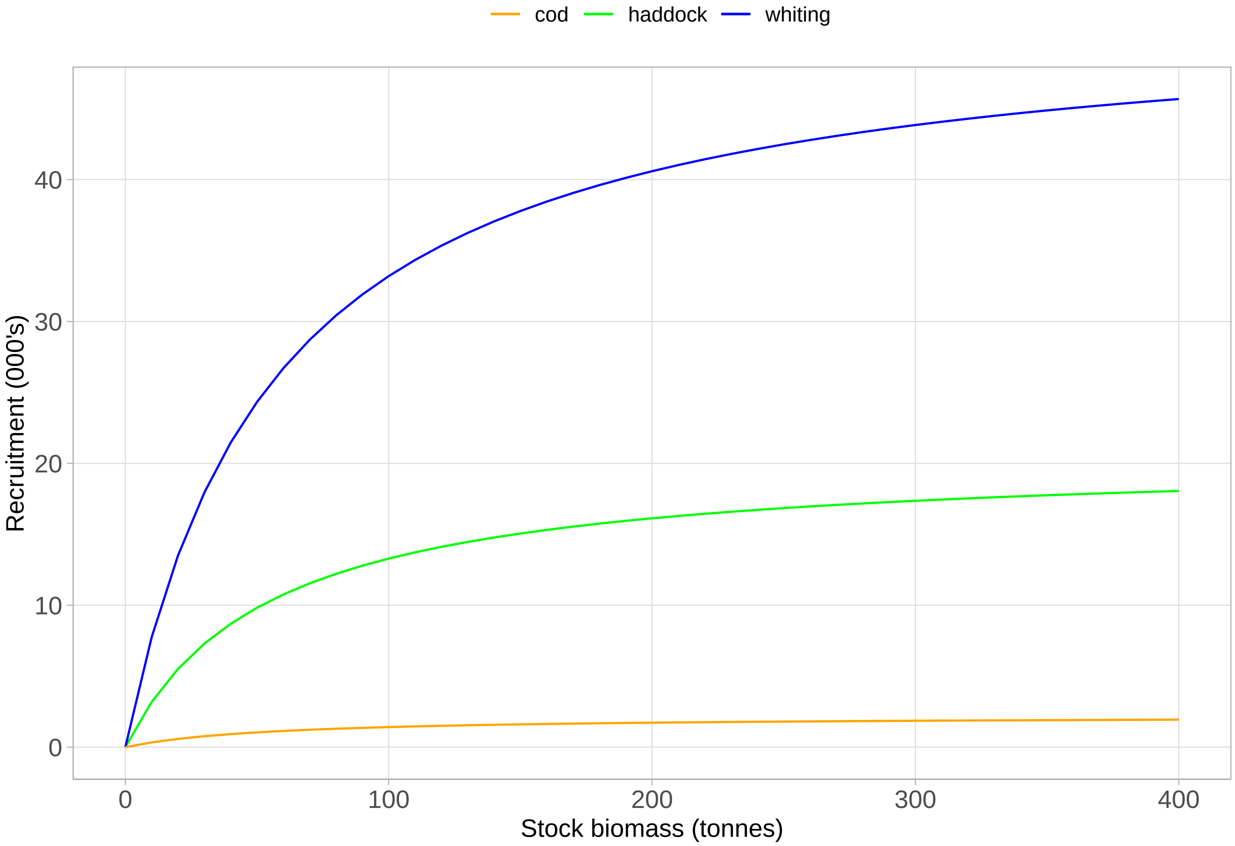


Figure F.8 - Stock biomass and recruitment curves (Beverton-Holt model) for cod, haddock and whiting.