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Using alternative biological information in stock assessment: condition-corrected natural mortality of Eastern Baltic cod

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Abstract

The inclusion of biological and ecological aspects in the assessment of fish population status is one of the bases for an ecosystem-based fisheries management. During the past two decades the Eastern Baltic cod has experienced a drastic reduction in growth and body condition that may have affected its survival. We used results from published experimental literature linking cod condition to starvation and mortality, to estimate the annual proportion of cod close to the lethal condition level in the Eastern Baltic cod stock. Thereafter we applied these results to adjust the natural mortality (M) assumed in the analytical stock assessment model. The results in terms of Spawning Stock Biomass (SSB), Fishing mortality (F) and Recruitment (R) in the final year from the stock assessment using M values adjusted for low condition were up to 40% different compared to the assessment assuming a constant M = 0.2. This method could be used for adjusting natural mortalities for other cod stocks where changes in condition are observed.

Keywords: ecosystem-based fisheries management, fish nutritional status, natural mortality, stock assessment

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Introduction

Ecosystem-based fisheries management relies on the use of biological and ecological considerations in the management of exploited resources (Pikitch *et al*., 2004; McLeod and Leslie, 2009). In practice, this can be achieved either by the use of ecosystem information directly as input in the assessment and/or forecast models (e.g. in form of climate-sensitive stock-recruitment relationships; ICES, 2015a) or by their integration in the management decision (e.g. environment-sensitive harvest control rules; Lindegren *et al*., 2009).

In the Baltic Sea, recent examples of the inclusion of ecosystem information in stock assessment are mainly confined to the pelagic stocks. Temperature and zooplankton are for example used in the short-term forecasts of the Gulf of Riga herring (ICES, 2015a), whereas the use of temperature has been attempted for Baltic sprat (ICES, 2007). Moreover, predation mortalities by cod from multi-species models are incorporated in the single-species stock assessment models for Baltic sprat and Central Baltic herring (ICES, 2015a). On the other hand, for the Baltic cod stocks there has been a lack of practical applications of ecosystem information into stock assessment. The only currently used application is the inclusion of cannibalism in the single-species stock assessment of the Western Baltic cod (ICES, 2015a).

During the past two decades, one of the main changes in the Eastern Baltic cod stock (hereafter simply referred to as the Baltic cod) has been the drastic decrease in body condition, and likely also growth, of individual fish (ICES 2015b; Eero *et al*., 2012, 2015). An association between poor condition/growth and natural mortality has been reported for a number of species in the wild and experiment setups (Adams *et al*., 1982; Henderson *et al*., 1988; Post and Evans, 1989; Thompson *et al*., 1991; Gislason *et al*., 2010). Experimental studies performed on Atlantic cod have also found a negative relationship between body condition and mortality (Dutil and Lambert, 2000), which the authors argued to be one of the causes of the lack of recovery of the Gulf of St. Lawrence stock despite the fishery moratorium in the 1990s. Based on literature, therefore, an increase in natural mortality could be expected also for the Baltic cod.

The aim of our paper was to use the findings by Dutil and Lambert (2000) to propose a method to estimate the annual proportion of the Baltic cod that is expected to die due to low condition, with an application into stock assessment. A comparison between the status of the stock (spawning stock biomass, fishing mortality and recruitment) estimated by assuming constant *vs*. condition-corrected natural mortalities is also presented using the settings of the latest accepted analytical assessment of this stock (ICES, 2013).

Methods

Data and estimation of body condition

Data on total length (TL, cm) and total weight (TW, g) of individual cod were extracted from the ICES DATRAS database, which contains biological information of cod caught during the Baltic International Trawl Survey (BITS; ICES, 2014a) and covers the period 1991-2014. Dutil and Lambert (2000) used gutted weights (GW) and fork length (FL) in their condition estimates, and therefore TL and TW were converted into GW and FL for each fish.

The formulas used for the conversions are the following:

$FL = 0.9306 * TL + 3.7471$ (eq. 1)

$GW = TW / (1.10 \text{ to } 1.23, \text{length-class and quarter-specific})$ (eq. 2)

The equation (1) was based on Pinhorn (1969), while the equation (2) was based on samples from the Baltic Sea collected by Sweden and Denmark in ICES SDs 25 and 26 between 2006-2014 (totally 21 436 cod). No differences in the weight conversion formula were observed between years and therefore an overall average by quarter and length-class was used. After conversion, the condition for each fish in the DATRAS database was estimated as Fulton's $K = GW / (FL\char`^3) * 100$.

Estimation of K-corrected natural mortalities (IBIS)

Dutil and Lambert (2000) carried out a controlled experiment with cod (in the length range of 30-55 cm) in which one group of fish was fed and one group was not fed for 92-100 days. While some fish died from starvation during the experiment, it was found that of the fish that were alive at the end of the experiment, the fish that were not fed (starved) had K=0.42-0.67 and the fed fish had K=0.83-0.98 (Fulton's K, based on fork length and gutted weight). The biological properties (e.g. liver energy and muscle energy) of some of the starved fish were similar to those of dead fish, so the authors expected them to die shortly.

The following steps were followed to estimate the proportion of fish that would die due to low condition and the new corresponding natural mortalities for wild Baltic cod:

1) Fish with $K \leq 0.65$ were considered as dying. The proportion of fish with $K \leq 0.65$ was estimated for BITS Q1 and BITS Q4 surveys between 1991-2014. The estimation was done by length-classes (20-29, 30-39, 40-49, 50-59 and 60-69 cm).

2) The proportions of fish with $K \leq 0.65$, by quarter and year, were translated into natural mortality rates (M_K) using the following equation (Nygård and Lassen, 1997):
"proportion fish with $K \leq 0.65$" $= (1-(\exp(-M_K)))*100$ (eq. 3)

3) The estimated natural mortality rates were added on the top of the constant M=0.2, traditionally assumed in the analytical assessment of the Baltic cod stock. By doing this, we made the assumption that the mortality = 0.2 is due to everything but condition. Therefore, the new M was estimated using the following equation:

$$M\ (\text{K-corrected}) = 0.2 + M_K \qquad (\text{eq. } 4)$$

4) The K-corrected natural mortalities by length were translated into natural mortalities by age using an age-length-key in which ageing was done using only one country, i.e. Sweden. The use of only Swedish age estimations in the age-length-key would reduce the effect of the inconsistencies existing in Baltic cod age estimations among countries (ICES, 2015b).

The flowchart of the steps used in the estimation of the K-corrected natural mortalities is shown in Fig. 1.

Application of the K-corrected natural mortalities into stock assessment

We used the K-corrected natural mortalities as input in analytical stock assessment using SAM (ICES, 2013). Analytical stock assessment has not been used in practice for this stock since 2014 due to a number of issues with input data and in the model outputs (ICES, 2015b; Eero *et al.*, 2015). Our purpose here was not to produce a functioning stock assessment to be used as basis for ICES advice, but rather to illustrate how much K-corrected natural mortalities alone would change the perception of the stock in relative terms, regardless of the other uncertainties that might be present in the assessment.

Therefore, we applied the same settings and other input data as used in the latest accepted assessment (ICES, 2013) and only changed the natural mortality values. We compared the output (spawning stock biomass, SSB, fishing mortality, F, and recruitment, R) of the runs in which natural mortalities were set to 0.2 for all ages (ICES, 2013) with runs in which K-corrected natural mortalities estimated following our method were used.

We first looked at the effect of modified M values in 1991-2014 on the entire corresponding time series of SSB, F and R from stock assessment up to 2014. Secondly, we were interested in demonstrating the effect of M values on SSB, F and R in the final year of stock assessment, which is the basis for providing management advice. Therefore, we successively excluded one last year from the time series back to 2004 (known also as retrospective analyses), both in the assessment with constant M values and the one with K-corrected Ms, to be able to compare the effect of M on the estimates in the final year in eleven cases (2004-2014). Thereafter we plotted the values of SSB, F and R for the final year of these eleven runs to show the differences between using constant M and K-corrected M values.

To demonstrate the potential effect of changes in M on management targets, maximum sustainable yield reference points (Fmsy and MSY) were calculated both for the assessment with constant M (at 0.2) and with K-corrected M, using the standard method used in ICES (e.g. ICES, 2014b).

Results

A total of 244 258 fish records were extracted from the DATRAS database. An example of frequency distributions of K-values by quarter is shown in Fig. 2. While in 1994 nearly no fish had K ≤ 0.65, in 2010 a significant proportion of fish was below this threshold in both quarters. The time-series of the proportion of fish with K ≤ 0.65 by length-class and quarter is shown in Fig. 3. The smaller length-classes (20-29 and 30-39 cm) did not show large changes in any of the two quarters, while the larger length-classes showed an increase up to 15% in quarter 1 and 25% in quarter 4. The proportion of fish with K ≤ 0.65 decreased towards the end of the time-series in quarter 4. The time-series of K-corrected natural mortality rates by quarter and length-class are also shown in Fig. 3, and display the same patterns as the proportion of fish with K ≤ 0.65. The time series of K-corrected natural mortality rate by age, estimated using a Swedish age-length-keys, are shown in Table 1.

Spawning Stock Biomass (SSB), Fishing mortality (F) and Recruitment (R) for the final year using the constant natural mortalities (i.e. 0.2; ICES, 2013) and the K-corrected natural mortalities are shown in Fig. 4. The differences between the runs ranged between 2-24% in SSB, between 3-40% in F, and between 2-14% in R in the eleven years compared. SSB and R were always higher and F was always lower in the runs with the K-corrected natural mortalities, with increasing difference over time due to cumulative effect of a higher M compared to the baseline value of 0.2.

The K-corrected M values resulted in ~ 40% higher Fmsy estimate compared to the Fmsy estimated using constant M = 0.2 (in our example, Fmsy were estimated at 0.63 and 0.43, respectively). Less difference was obtained for MSY, the value corresponding to K-corrected M being about 5% lower than for constant low M (60430 tons and 63516 tons, respectively).

Discussion

One of the ways to foster a sustainable ecosystem-based management is to integrate biological and ecological knowledge into stock assessment and management of the exploited living resources.

In terms of natural mortality, the ecosystem effects that are considered in stock assessment context are mainly related to predation (e.g.; Sparholt, 1990; Hollowed *et al*., 2000; Tyrrell *et al*., 2008; EC, 2012). Age- or size-specific differences in natural mortality are in some cases assumed or estimated using length or weight data (e.g. for plaice in Eastern Channel; ICES, 2015c). However, we are not aware of any examples where inter-annual variations in natural mortality due to other drivers than predation have been formally included in fish stock assessments. This is likely because natural mortality is generally difficult to estimate, which is why it is often kept constant (Johnson *et al*., 2014). Nevertheless, in cases where strong time trends exist in some biological or ecosystem parameters that are expected to influence natural mortality, efforts should be made to take this into account in stock assessment.

In the Baltic Sea, the drastic decrease in condition of cod during the past 2 decades has raised a lot of concerns for the managers and the fishery (ICES, 2015a), including decrease in the quality of the fish and consequently their prices, possibly contributing to increased discarding and high-grading and to the impossibility to fill the annual quotas. The reasons of the observed decrease in cod condition are outside the scope of our paper and are currently under investigation in several projects. Recently published papers and reports point to the lack of food and density dependence (Eero *et al*., 2012), selective fishing (Svedäng and Hornborg, 2014), increase in anoxic areas and increase in parasite infection (ICES, 2015b) as potential causes for the decrease in cod condition.

Our study aimed at proposing a method to include information about the temporal variations in cod condition in the estimation of cod natural mortality, to be applicable in stock assessment. Analytical stock assessment has been used by ICES as base for Advice for the Eastern Baltic cod up to 2013 (ICES, 2105b). Thereafter, due to issues discovered in the analytical assessment (e.g. strong retrospective patterns), it was abandoned and a data-poor approach (based on biomass trends from scientific surveys) was used as base for advice (Eero *et al*., 2015). The potential reasons of the failure of analytical stock assessment have been summarized by Eero *et al.* (2015) and are represented by deteriorated ageing quality, unaccounted natural mortality, uncertain discards and changes in survey catchability. Therefore, our study represents an attempt to solve some of these issues. Moreover, our study illustrates how much the M-correction alone would matter in the perception of the stock in relative terms, regardless of all the other uncertainties that exist in the assessment.

Consequently, the results in terms of SSB, F and R presented in this study are only indicative and should not be used for management purposes. Our proposed method to estimate K-corrected natural mortalities can be applied (both in age- and length-based models) when the other issues related to the analytical stock assessment will be solved.

Recommendations for fisheries management in terms of fishing mortality (F) often scale with natural mortality (M), and may therefore be highly dependent on the value of M assumed (Andrews and Mangel, 2012). Thus, misspecification of M may lead to over- or under-estimates of critical management quantities (Johnson *et al.*, 2015). In our analyses, not taking into account the increase in M due to low condition resulted in lower SSB estimates and higher F estimates compared to the analyses that included K-corrected M. Thus, in this particular example and present situation, one may argue that ignoring the increase in M and applying a constant relatively low value of M would be more precautious from a management perspective, since it would decrease the catch advice. However, this could be different in a situation when M is changing in the other direction, e.g. if the condition of cod will improve. Furthermore, setting targets for fishing mortality at increasing M is not straightforward and two opposing possibilities may be considered. One approach would increase the target fishing mortality based on yield per recruit considerations, as the change in FMSY owing to changing M in our study would also suggest. In contrast, another approach would lower the target F to maintain the total amount of mortality the stock can withstand before productivity is impacted constant (Legault and Palmer, 2015). In this context, Rätz *et al*. (2015) introduced the concept of maximum sustainable dead biomass (MSDB) as a way to account for variations in natural mortality and fishing mortality over time in management. The MSDB approach would provide managers with the level of different natural mortality sources and adjust the anthropogenic mortality accordingly.

In practice, thorough analyses would be needed to determine the most appropriate approach via estimating targets for a range of stock–recruitment relationships and evaluating the trade-offs between risk of overfishing and lost yield (Legault and Palmer, 2015). Our aim in this paper was not to conduct such analyses, given the present lack of analytical assessment for the Eastern Baltic cod, implying that also reference points cannot be reliably estimated. However, we wish to point out that if time-varying natural mortalities are implemented in the future, the estimation of the corresponding management targets would need careful consideration.

In our analysis we focused solely on the impact of condition on natural mortality, to specifically look at the effect of one specific factor on the assessment results. However, other parameters used as input in stock assessment can be potentially affected by the condition of the fish. Fish in low condition have a lower probability of being mature (Morgan, 2004) and therefore a population dominated by slim fish is expected to have a lower SSB and recruitment success (Balcombe *et al.*, 2013). These effects of condition on maturation, SSB and recruitment could change the stock-recruitment relationship and affect the outcome of the stock assessment and estimation of reference points. Moreover, the occurrence of slim fish in the stock could be expected to change the selectivity of both commercial and scientific surveys (ICES, 2015a).

Hereafter, we discuss our assumptions and the potential shortcomings of the method we have proposed. Firstly, we assumed that the cod with $K \leq 0.65$ are all dying, while in the experiment performed by Dutil and Lambert (2000) non-fed fish with $K \leq 0.65$ either died from starvation or were still alive at the end of the experiment. However, the biological properties (e.g. liver and muscle energy) of the starved fish still alive resembled that of fish that died, and therefore the authors expected the starving fish to die shortly. In the wild, fish experience harsher environmental condition than in controlled experiments. Especially, cod in the Baltic Sea is at the limit of its distribution range, and is heavily affected by environmental stressors such as large variations in the already low salinity and large extent of hypoxic areas.

Moreover, starved fish in the wild could be more prone to predation and fishing as well as have increased difficulties in feeding on highly motile pelagic prey. Therefore, we consider reasonable our assumption that all fish with $K \leq 0.65$ would die. Secondly, Dutil and Lambert (2000) used in their experiment fish in the size range 30-55 cm, and therefore the limits for mortality and starvation for the other size-classes are unknown. The size-class 20-29 cm used in our calculations for the Baltic cod presents very little fish with $K \leq 0.65$ (around 1%) and therefore a bias for those would produce only a very minor difference in our calculations. On the other hand, although it is hard to believe that cod in the size-class 60-69 cm would have a mortality (or starvation) limit very different from the one used in Dutil and Lambert (2000), this could have been a source of bias. Thirdly, in our survey the cod that might have already died due to low condition are of course not observed. This means that we could have underestimated the proportion of the population that die because of low condition, or could potentially compensate for the first assumption. Another source of uncertainty is associated with the laboratory experiments originating from the northern Gulf of St. Lawrence (Dutil and Lambert, 2000), while the thresholds for detrimental condition may be different in other cod stocks. For example, the experiments were conducted at higher temperatures than those normally experienced by the Baltic cod that could affect the threshold level of mortality due to low condition. Fourthly, the length-age-key (ALK) we used to allocate length-classes into ages can contain bias due to the known ageing problems for the Baltic cod. The use of only Swedish ALK, although potentially being also inaccurate, would reduce these temporal inconsistences and bias. Fifthly, we assumed that the mortality due to low condition has to be added to the background natural mortality of 0.2, which could be caused by diseases, predation by seals and cannibalism for smaller individuals. In the 1990s there was nearly no cod with $K \leq 0.65$, nevertheless an $M = 0.2$ was used in analytical stock assessment (ICES, 2013). This supports our choice to add the mortality due to low condition to 0.2. Finally, various mortality effects may be compensatory, i.e. they may decline as other effects increase. Natural mortality can either compensate for the anthropogenic mortality (i.e. decrease of the natural mortality in exploited systems) or overcompensate for it (i.e. increase of the natural mortality in exploited systems). If the exploitation is targeting individuals with lower than average quality, the mortality from exploitation could be compensated by a decreased natural mortality. Alternatively, if exploitation is targeting individuals with higher than average quality, exploitation could be over-compensated by increased natural mortality since the remaining individuals would be below average and thus more susceptible to die of natural causes, as shown in both aquatic and terrestrial systems (Toigo *et al*., 2007; Sandercock *et al*., 2011; Sedinger *et al*., 2010; Rätz *et al*., 2015). What are the main mechanisms acting on the Baltic cod remain an open question. It could be argued that trawling would be compensatory since cod with lower condition is more easily caught than cod with a higher condition, leaving cod with good condition left in the system. Conversely, slim cod could actually escape through the mesh of the trawl, proportionally removing cod in good condition from the system.

We have proposed here a simple approach that can be considered to adjust the natural mortalities from body condition data for stock assessment and management purposes, using available information from existing experimental studies. We recognize that ideally a validation of the resulting estimates for a given stock would be needed. However, at lack of such validation, existing information from other stocks can be useful to explore the potential consequences of changing M, also in the context of defining precautionary management targets. Therefore, we consider the approach useful both for Baltic cod and for other cod stocks, especially those experiencing large fluctuations in condition and growth.

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Figures

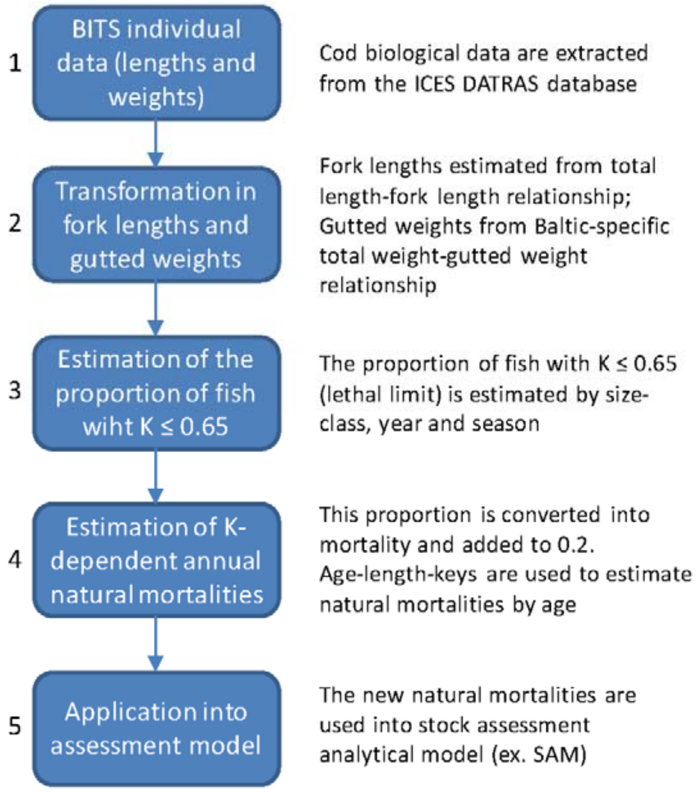

Figure 1. Flowchart of the steps used in the estimation of the K-corrected natural mortalities for inclusion in analytical stock assessment models. In step 4, the translation of natural

mortalities by length-classes into natural mortalities by ages is not necessary if a length-based stock assessment model is used.

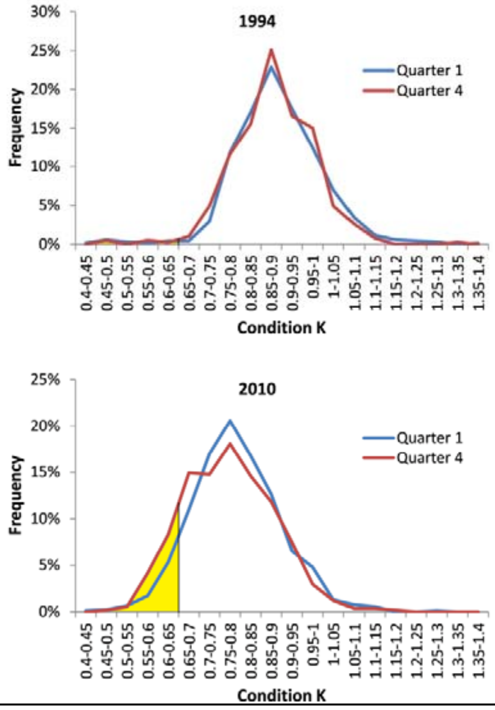

Figure 2. Examples of frequency distributions of condition (1994 and 2010) of cod in the size-class 50-59 cm, by quarter. The shaded area indicates the part of the distribution that is ≤ 0.65 and that was assumed to die.

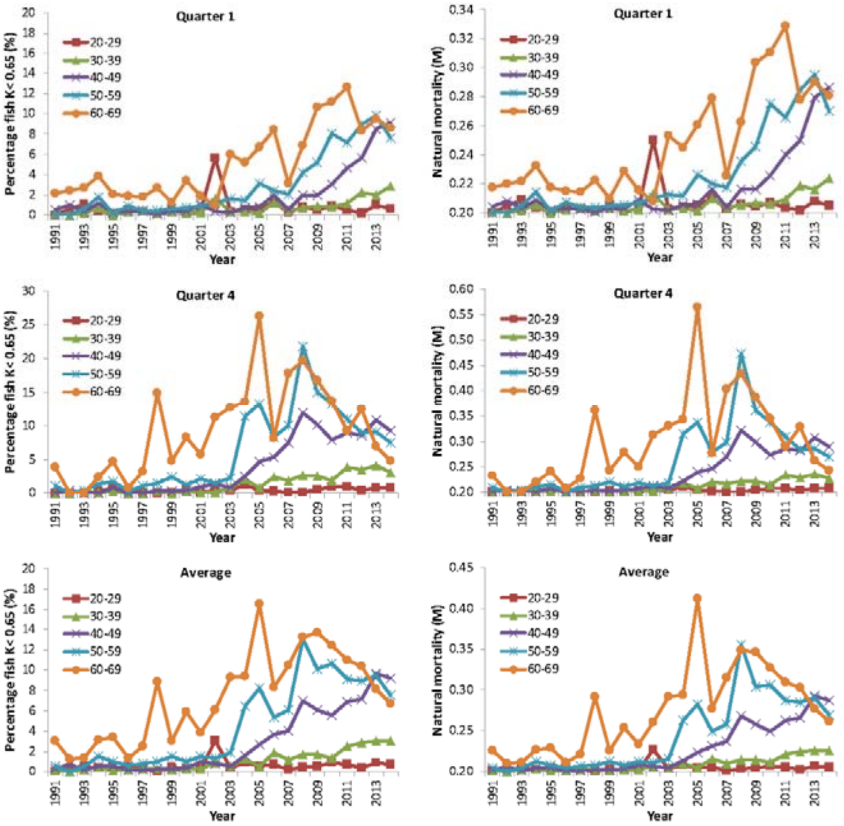

Figure 3. Time-series of condition (left panel) and condition-corrected natural mortality (right panel) for different size-classes of cod, by quarter. Averages between quarters are also shown. Note the different scale of the Y-axes.

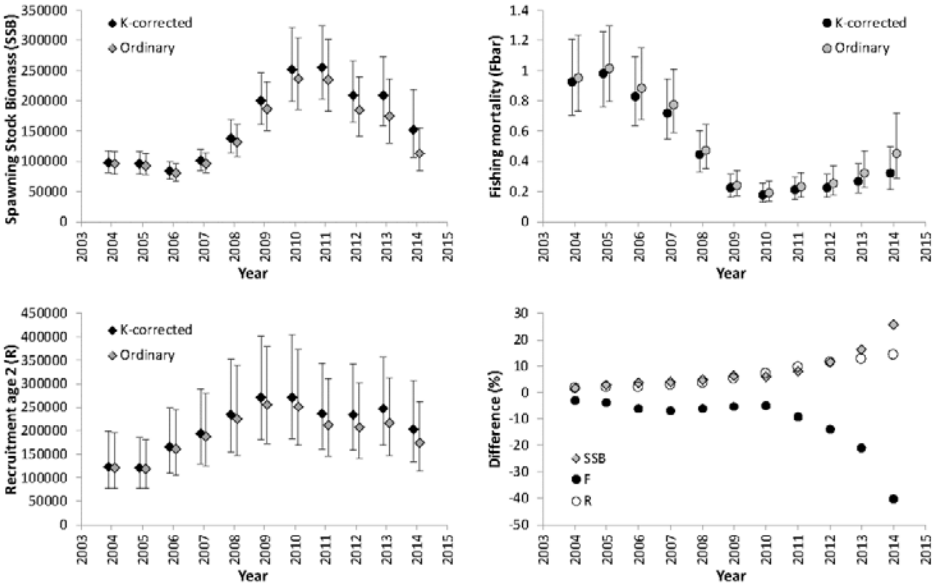

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Figure 4. Comparisons of Spawning Stock Biomass (SSB), Fishing mortality (F) and Recruitment (R) by year for the SAM runs with constant natural mortality of 0.2 (as regularly used for this stock) and the SAM runs with K-corrected natural mortalities. The symbols represent the final year estimate of 11 runs where we excluded step-by-step one year at a time back to 2004 (i.e. they represent the final year of 11 retrospective runs ending from 2004 to 2014). The relative differences in SSB, F and R using SAM runs with constant natural mortality of 0.2 and the SAM runs with K-corrected natural mortalities are also shown. Bars show 95% confidence intervals.