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– a project synthesis

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Developing management goals and associated assessment methods for Sweden’s nationally managed fish stocks – a project synthesis

Utveckling av förvaltningsmål och tillhörande bedömningsmetoder för nationellt förvaltade arter – en projektsyntes

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Summary

This report summarizes and synthesizes results from the Swedish Agency of Marine and Water Management (SwAM, or HaV) funded project "Förvaltningsmål för nationella arter (Management goals for nationally managed species)". The objectives of the project have been to promote the development of management goals and associated status assessment methods and indicators, as well as reference points, for some nationally managed fish stocks both in coastal as well as freshwater areas. The report focusses largely on species and stocks that can be defined as data-poor. Such stocks are characterised by marked limitations in data availability and/or resources allocated to detailed analytical stock projections. Data-poor stocks also often lack carefully formulated management goals and associated methods and indicators for assessing stock status. In this report, we provide an overview of potential assessment methods and indicators and try to synthesise how they work and what the strengths and weaknesses are by applying them to selected data poor stocks such as pikeperch, pike, whitefish, and vendace. We also discuss how they relate to different potential management goals and provide recommendations for their application. We grouped the indicators and assessment methods by the three categories that are now used in the yearly status assessment framework provided by SLU Aqua (Resursöversikten/Fiskbarometern) – i) mortality, ii) abundance/biomass and iii) size/age structure. The results are also described for these three main categories of assessment indicators. Included is also a status report from a size- and age-based population dynamics model (Stock Synthesis 3) that is being developed for pikeperch in Lake Hjälmaren.

An important experience from the project is that to improve the assessment methods for Swedish national fish stocks, it is important that managers develop both general as well as more detailed quantitative goals for the individual stocks. This should ideally be conducted in various forms of collaboration with the main stakeholders and scientists involved with assessment as participatory processes foster legitimacy. Carefully articulated management goals, which are possible to translate into quantitative targets, will facilitate the development of various approaches and methods to monitor stock statuses. Given the strong and complex interactions of fish and their environments it is also important to consider other pressures than fisheries when developing indicators and assessment methods.

Our synthesis highlights a number of areas where the assessment of data-poor stocks can be improved:

1. **Apply precautionary principles** for data-limited stocks, particularly ones that are known to be vulnerable to exploitation.
2. **Tailor approaches to how fisheries are managed in Sweden**. Swedish nationally managed fish stocks are not managed by quotas (with one exception, vendace in the Bothnian Bay) and do not aim for maximum sustainable yield. Instead, the coastal and inland fisheries are managed by regulating the effort in the small-scale commercial fisheries (number of fishers/licenses and amount of gear). Regulation of recreational and subsistence fisheries effort, in terms of licenses or number of fishers) is not applied, nor possible since the fisheries is lacking obligatory notification and reporting systems. All national fisheries, however, are regulated by various technical measures (closed areas, size-limits, bag-limits, gear restrictions etc). Thus, goals and assessment methods that result in harvest limits or quota recommendations expressed in e.g. biomass/numbers are difficult to use as basis for management. Instead, there is a need for alternative management goals and associated assessment methods.

3. **Use best practice** methods and indicators and adapt as scientific knowledge is developed. Data-limited methods are developing rapidly, and new methods/approaches are proposed in the scientific literature every year. It is thus important to be updated on the most recent developments.
4. **Clearly describe limitations/assumptions** of methods used. It is important to be aware of and critically evaluate the assumptions underlying the analyses, and to carefully communicate uncertainty together with the stock status assessment.
5. **Be particularly careful with low sample numbers**. Many indicators and methods can be applied also on small sample sizes, however, the accuracy and precision of the estimates risk being low in such cases.
6. Accept that there is no "gold standard" for fisheries assessment. **Each case study is unique and needs to be balanced against data availability, local needs and other important factors**. This also means that analysts need to be careful when using generic reference levels or "borrowing" data from other stocks.
7. **If possible, use several different methods/indicators**. Although several indicators aim to measure similar aspects of the stock, small methodological differences can support the overall interpretation of individual indicator values. It is particularly important to incorporate many aspects and indicators (size/age/abundance/mortality) in order to produce a balanced assessment.
8. **Develop means of communication.** Indicators and goals should be easy to understand. However, interpretation of results from multi-indicator frameworks can be challenging. There is thus a need for finding ways of communication that can convey complicated results in a simple-to-understand manner.
9. For details on additional improvements, we refer the reader to the sub-header "recommendations for the future" found under each chapter.

The implementation of Stock Synthesis for pikeperch in Lake Hjälmaren showed that it is possible to develop a more ambitious and detailed stock assessment model for a relatively data-poor stock. The model results partly support earlier interpretations of the development of the stock and the importance of the changes in regulations in 2001 (increased minimum size, increased mesh size and reduced mortality of undersized pikeperch). Before the model can be implemented and used for practical management, a number of actions for improvement are needed, which are highlighted in the relevant chapter. The most important next step is establishing management goals and reference levels for this stock. We recommend that such a dialogue is initiated by managers. The fisheries management goals should consider both biomass, fisheries mortality and size-based targets.

To conclude, we stress the importance of improving all ongoing aspects related to the assessments of data-poor Swedish stocks. Strong local stocks and sustainable fisheries are vital for a variety of fisheries-related businesses and practices, particularly in rural areas, providing economical and societal value. Fishes also have important roles in aquatic food-webs and it is important that ecological values are managed wisely in order to reach targets for water quality, ecosystem structure and diversity. Given the strong and complex interactions of fish and their environments it is also important to consider other pressures than fisheries when developing indicators and assessment methods.

Sammanfattning

Denna rapport sammanfattar och analyserar resultat från det HaV-finansierade projektet "Förvaltningsmål för nationella arter". Projektet syftar till att bidra till utvecklingen av

förvaltningsmål och tillhörande beståndsanalyser, referensnivåer och indikatorer för svenska nationellt förvaltade bestånd längs kusten och i de stora sjöarna. De flesta av de aktuella bestånden är så kallade data-fattiga bestånd. Med det menas att det finns brister i datainsamlingen och att det inte finns tillräckliga resurser för att genomföra detaljerade och fördjupade analytiska beståndsanalyser. För de flesta av dessa bestånd saknas också tydliga mål för förvaltningen och således inte heller indikatorer/analyser som kan användas för att följa upp målen. Resultaten i projektet sammanfattas för tre huvudsakliga kategorier av indikatorer/analysmetoder: dödlighet, storleks- och åldersstruktur samt abundansmått. Det finns också ett kapitel som beskriver arbetet med att utveckla en storleks- och åldersstrukturerad populationsmodell (Stock Synthesis 3) för gösbeståndet i Hjälmaren. En viktig erfarenhet från projektet är att förvaltande myndigheter behöver lägga särskilt fokus på att formulera förvaltningsmål. Dessa mål bör vara uppföljningsbara. Det är också viktigt att målen formuleras i samverkan med berörda fiskeintressenter och forskare eftersom deltagandeprocesser ökar legitimiteten. Välformulerade, väl förankrade och genomtänkta förvaltningsmål är en förutsättning för att kunna fortsätta utveckla analysmetoder och indikatorer för att bedöma de nationella beståndens status.

Vår utvärdering visar att det finns visa områden som är särskilt viktiga att beakta om förvaltningen och analyserna för nationella fiskbestånd ska förbättras. Dessa är:

1. **Använd försiktighetsansatser för de bestånd som är särskilt fattiga på data**. Detta är extra viktigt för bestånd som är särskilt sårbara för exploatering.
2. **Anpassa metoder och arbetssätt till de förutsättningar som finns** för svenska nationellt förvaltade bestånd. Fångstkvoter för fisket tillämpas inte för nationella bestånd, med ett undantag, siklöjan i Bottenviken. Istället förvaltas fisket nationellt genom att ansträngningen i yrkesfisket regleras (antal fiskare/licenser samt dispenser för redskapsansträngningen). Eftersom det saknas anmälnings- (och rapporterings)skyldigheter inom fritidsfisket kan ansträngningen i detta fiske inte regleras lika lätt. Allt fiske, inklusive fritidsfiske, reglerasdockgenom att man använder tekniska regler som exempelvis bag-limit, minimimått, fredningsområden och redskapsbegränsningar. Således är det svårt att direkt tillämpa mål och metoder som endast omfattar/resulterar i förslag på kvoter inom förvaltningen. Istället behövs alternativa formuleringar av mål och metoder för uppföljning.
3. **Använd väl beprövad metodik och tillhörande indikatorer**. Det pågår en hel del forskning om hur man kan utveckla beståndsanalyser för datafattiga arter och därför sker just nu en snabb utveckling av metoder och nya indikatorer föreslås varje år. Det är därför viktigt att både målformuleringar och uppföljning kan uppgraderas med jämna mellanrum för att så långt möjligt anpassa sig till kunskapsläget.
4. **Beskriv tydligt de olika metodernas begränsningar, osäkerhet och antaganden**. Det är viktigt att de brister och osäkerheter som finns i många analyser kommuniceras tydligt och att resultaten alltid granskas kritiskt innan de publiceras och används inom förvaltningen.
5. **Var särskilt försiktig i situationer när provstorleken är liten**. Många indikatorer och analyser kan användas trots att man har ett litet antal prover att utgå ifrån. Det man dock behöver vara medveten om är att både precision och exakthet kan påverkas negativt och att det i värsta fall finns en risk att man drar felaktiga slutsatser.
6. **Acceptera att det inte finns en universell standard för beståndsanalyser**. Varje situation och bestånd är unikt, vilket innebär att analyserna alltid måste inkludera en balanserad avvägning som tar hänsyn till vilka data som finns tillgängliga, vilka förutsättningar som finns och vilka lokala önskemål som finns inom förvaltningen. Det är

således särskilt viktigt att vara försiktig om man ”lånar” data från närbesläktade arter, närliggande bestånd eller likartade platser.

7. **Om möjligt använd många olika metoder och indikatorer**. Det finns i vissa fall många snarlika indikatorer som syftar till att bedöma ungefär samma saker. Trots att skillnaderna mellan indikatorer ibland kan upplevas som hårfin kan det finnas relevanta skillnader i struktur och uppbyggnad. Därför kan det vara värdefullt att testa flera olika indikatorer för att kunna jämföra utfallet för ett givet bestånd. Det kan vara särskilt viktigt att testa indikatorer som baseras på <u>olika</u> typer av information (exempelvis dödlighet, ålder, storlek och/eller abundans/biomassa). Genom att jämföra resultat från flera olika indikatorer kan man göra en mer samlad bedömning av beståndets status.
8. **Utveckla kommunikationen om förvaltningsmål och metoderna för uppföljning**. Såväl indikatorer som förvaltningsmål ska helst vara tydliga och enkla att förstå. Tyvärr kan vissa mer komplicerade analyser och tillhörande indikatorer istället vara svåra att förstå. Ännu mer komplext kan vara de situationer där man väger samman resultatet från många olika indikatorer. I takt med att beståndsanalyser blir mer och mer komplicerade finns således ett stort behov av att kunna utveckla hur de kan kommuniceras och förstås även för en bredare allmänhet som saknar särskilda förkunskaper.
9. För mer detaljer om andra förslag på förbättring hänvisas till särskilda avsnitt i varje kapitel som berör just rekommendationer för framtiden.

Sammanfattningsvis visar resultaten från projektet på betydelsen av att förbättra förvaltningsmål och beståndsanalyser för datafattiga svenska fiskbestånd. Fisket är måhända en rätt så liten bransch men i vissa områden på landsbygden kan fiskerirelaterade företag ha en stor betydelse. Bättre förvaltning av våra nationella fiskbestånd är heller inte endast av betydelse utifrån ekonomiska aspekter, det är även värdefullt ur sociala och ekologiska perspektiv. Fisken och fisket har också en stor betydelse för att ekosystemen i våra sjöar och kustområden ska vara välmående och för att kunna klara övergripande målsättningar för ekosystem och vattenkvalitet.

Contents

1. Introduction

Improving the scientific basis for management of data-limited stocks has become a major challenge in fisheries biology. Data-poor situations are rather common all over the world and a majority of the stocks across the globe lack formal status assessments (Costello et al. 2012; Ovando et al. 2021). Data-poor situations occur particularly for species that are important for recreational fisheries, species without or with less significant commercial value and/or for species that are not caught with standard monitoring gears. The resulting uncertainty in the precision and adequacy of stock assessments may lead to inadequate management measures or even stock depletion (Hilborn et al. 2020).

Sweden has a long coastline, many large lakes and rivers and thus have a bigger challenge with data-poor situations than many other places in Europe. There are several fish stocks within the five large lakes in Sweden and along the coast that can be considered data-limited (Östman et al. 2016) and the status of these stocks is often uncertain (HaV 2022). Some of these stocks also have a complex population structure and one lake or coastal area can have several populations of a species although the exact kinship and demography of the populations are seldom known. The term data-limited, or data-poor, is a broad term that is meant to describe stocks for which there is limitations as regards the assessment of the status. It includes both situations that are limited by data, e.g. in terms of quantity, quality or type of data, as well as situations that are related to resource-limitation, e.g. in terms of analytical technical capacity, financial support for data collection, analyses and time (Cope et al. 2023). Nevertheless, in many cases, there are limited possibilities for gathering the in-depth data required for advanced stock analyses. As a result, there is (in the broad sense) a need for assessment methods and indicator frameworks that are better adapted to data-limited situations. Such analyses will enhance the potential to assess stock status and to support the development of management targets of data-poor stocks (i.e. the ideal stock status given local considerations and data limitations).

In this Aqua Report we report on work conducted within the project "Förvaltningsmål nationellt förvaltade arter" (Management targets for nationally managed species), financed by Havs- och vattenmyndigheten (HaV, dnr 896-20). The overarching purpose of the project has been to promote the development of management objectives and targets, as well as associated reference points, for nationally managed species and stocks. From the perspective of SLU, the project has thus worked as a stimulus for the development, test and adaptation of assessment methods and indicator frameworks for Swedish conditions. The results developed within the project has then, where appropriate, been implemented in the status assessment work conducted within related projects. The project has also provided direct support to both national and regional fisheries management in Sweden, e.g. by participating in several management processes and assisting in the on-going revision of the fisheries management plan in Lake Vättern (Vätternvårdsförbundet 2017) and the up-coming plans in L. Mälaren and L. Hjälmaren. The practical work has been a continuation of earlier versions of the project (Florin et al. 2017; (Östman et al. 2016).

Östman et al. (2016) reviewed available management targets and associated methods for assessing whether fisheries exploitation levels of nationally managed fish stocks were sustainable. They provided a summary of various possible quantitative management targets and methods, with a focus on indicators of stock status in the absence of quantitative stock models. In addition, they assessed the type of data required for the different methods and management goals, as well as what data are available. The report notes that "Implementing quantitative stock models to assess the status of all exploited stocks in Sweden is not, and will never be, possible". Instead, in many cases, simpler indicators that reflect a stock's status must be applied." For most nationally managed stocks, data are generally too limited to propose specific targets, that instead of relying on a single target/indicator managers should use several different management targets or indicators of exploitation to most effectively facilitate sustainable use of aquatic resources. The preliminary goal of this project (which is synthesised in this report), was to identify potentially suitable methods of assessing data-limited species in Swedish waters and test these methods on a subset of species. Several stocks have been identified as data-limited, for example pike (gädda - *Esox lucius*), whitefish (sik - *Coregonus lavaretus*), pikeperch (gös - *Sander lucioperca*) and vendace (siklöja - *Coregonus albula*). The early development period of the project was spent finding and assessing quality of available historical data for these species, identifying current knowledge, gaps in data and assessing the suitability of several potential assessment methods based on available data. As such, a range of indicator and assessment methods have been tentatively applied on several stocks in both coastal and lake areas. In this report, we provide an overview of potential indicators and methods but do not show the application to all the tested stocks. We instead try to synthesise how they work and what the strengths and weaknesses are by applying them to selected stocks. We also discuss how they relate to different potential management goals and provide recommendations for their application. More specifically, the aims of this report are, in relation to existing and potential management goals to:

1. Describe how different indicators and assessment methods work and the theory and assumptions behind them.
2. Use selected species and stocks and to provide readily and easily understandable outputs that clearly illustrate how they function operationally.
3. Discuss challenges and lessons learned.
4. Based on the above, provide recommendations for the future.

We do this by grouping the indicators and assessment methods by the three categories that are now used in the yearly status assessment framework provided by SLU Aqua (Resursöversikten/Fiskbarometern): i) mortality, ii) abundance/biomass and iii) size/age structure.

In addition, we have worked on two parallel parts of the project:

5. The development of a more advanced stock assessment model for pikeperch in Lake Hjälmaren.
6. The development of a framework for a fish recruitment indicator for coastal areas of the Baltic Sea.

The stock assessment model for pikeperch has not previously been published and is therefore included as a separate chapter in this report. To work towards an operational recruitment indicator, ensemble modelling approaches have been used to map the extent of recruitment habitats of a large set of fish species along the Swedish coast (Erlandsson et al. 2021; Fredriksson et al. 2021). Two approaches, focusing on habitat extent and fish abundance, that are related to the Marine Strategy Framework Directive (MSFD) criterion on the "necessary extent and condition to support the different stages in the life history of the species" (Descriptor 1 and Criteria 5 (D1C5)) have been examined. These recruitment habitats are today impacted by sticklebacks (Eklöf et al. 2020), and an attempt to assess the magnitude of this impact has also been included in the project (Bergström & Erlandsson 2022). The next steps of the work will involve further methodological development towards establishing operational area- and abundance-based state indicators for the extent and condition of fish recruitment habitats, and ultimately to define reference points and management targets.

2. Mortality related indicators

Mortality is a fundamental parameter in population dynamics of exploited fish stocks and estimates are commonly used as management targets. Mortality of exploited fishes is usually separated into two components: natural mortality (M), which includes inter- and intraspecific predation and all other natural causes, and fisheries related mortality (F), which can include mortality related to harvest and handling (bycatch, catch-and-release etc) in a fisheries. The total mortality is the sum of natural and fisheries mortality (Z=M+F). In data-poor stocks, however, natural mortality is an inherently difficult parameter to estimate (Maunder et al. 2023) and in data-poor stocks it is thus seldom known. An approximate estimate can be obtained from life-history invariants or, if available, be derived from studies on related stocks/species (see papers by Pauly 1980; Chen & Watanabe 1989; Lorenzen 1996; Gislason et al. 2010; Then et al. 2015, 2018; Thorson et al. 2017; Thorson 2020; Dureuil & Froese 2021; Hamel & Cope 2022; Lorenzen et al. 2022). Mortality can be expressed in several different ways. The annual mortality rate is the proportion of the total stock (in numbers) that die each year. The instantaneous mortality rate (Z) is the percentage of fish dying at any moment.

2.1. Formulation of objectives and targets

"Less than natural"

The idea that "humans should not take more than nature", i.e. using M as an upper limit for F under the concept of maximum sustainable yield has a long tradition in fisheries science (see e.g. references in Froese et al., 2016). They conclude that fisheries mortality should not exceed natural mortality and that half of natural mortality is a reasonable and long-term sustainable target for management:

"Fishing should be at or below Fmsy".

This is one of the targets of the EU:s common fisheries policy and also an important target for Swedish Agency of Marine and Water Management (SwAM), which should apply to the majority of Swedish fish stocks. Fmsy is the level of fisheries mortality that allows for the maximum level of compensation of the exploited stock. Thus, this level of fishing mortality will lead to maximum sustainable yield in the long term. Fmsy has received criticism for not properly taking ecosystem related constraints into account (Zhang & Megrey 2006; Sparholt et al. 2019).

"Total mortality (Z) should be equal or lower than a reference period".

This is an example of a regional target using mortality as an indicator, in this case from the Lake Vättern fisheries management plan (Vätternvårdsförbundet 2017). From a management perspective, this implies that fisheries induced mortality needs to be managed in relation to changes in natural mortality so that the sum (Z) remains constant or is reduced.

2.2. How do the indicators work

Total mortality in data-poor stocks is often calculated using various catch-curve analysis, which are reliant on information about the age and size structure of the catch or survey data. The most simple and straightforward way to calculate a catch curve is the regression-based method (Ricker 1975). This method has, however, not always performed very well in simulation studies compared to other methods (Smith et al. 2012). Instead, we have mainly used the Chapman and Robson estimator (Chapman & Robson 1960) or the method put forward by Millar (2014). A recent paper by Mainguy & Moral (2021) suggests another related approach to perform catch-curves. Examples of other methods are length/biomass cohort analysis (Jones 1990; Zhang & Megrey 2006), mark-recapture models (Miranda and Bettoli 2007) and analytical stock assessment models (Hamel et al. 2023). Natural mortality is inherently difficult to estimate, and is often inferred, or calculated, from various growth parameters (e.g. Hoenig, 1983; Jensen, 1996; Pauly, 1980). It also changes markedly as fishes grow. In early life-stages, mortality is often high, but already after a few years in life it decreases and starts to level out (Lorenzen et al. 2022). Depending on when fish are recruited to a fishery the average natural mortality could thus vary drastically depending on whether parts of the period with higher mortality is included or not.

Catch-curves uses age- and length frequency data to analyze to what extent the population density is receding with age. The basic idea is to focus on the negative correlation between density or a proxy of density (CPUE or number of individuals in the sample) and age. Application of a catch curve requires that certain conditions/assumptions are met. The main ones are: 1) steady state conditions with no marked changes in how the fisheries are operated during the study period, 2) unbiased measures of age and length structure in the population, 3) no trend in recruitment, preferably also low variation in recruitment, 4) mortality is relatively constant over time and within the selected age groups, (5) within the selected age/size interval individuals are equally available and vulnerable to the fishery (also applies to monitoring if such data are used).

2.3. Current use of indicator

Fisheries mortality is used as an important biological reference point for coastal vendace stocks (HaV 2022). In this specific case, mortality from seal predation is also accounted for and combined with fisheries related mortality. Fisheries mortality, estimated using analytical stock assessment models, has also been as an indicator of stock status for land-locked salmon of L. Vänern (Whitlock et al., in prep), vendace in L. Vänern (Ogonowski et al., in prep) and pikeperch in L. Hjälmaren (see later section on Stock Synthesis 3 models). Estimates of total mortality have been used in the assessment of coastal perch stocks, coastal flounder stocks, lake as well as coastal stocks of whitefish, Arctic char in L. Vättern and on pikeperch in lakes and coastal areas. Berggren et al. (2022) also calculated total mortality for Northern pike in L. Mälaren and coastal areas of the Baltic, which allowed a comparison of mortality in areas that are closed to fishing (i.e. F is assumed to be zero) with areas where fishing is allowed and also with areas with high natural predation from seals. Taken together, inferences on the importance of fisheries and natural mortality for stock status could be made.

Defining reference points or setting management goals regarding mortality can be difficult. Nevertheless, by comparing different management regimes or by monitoring development over time, management advice can still be produced as long as uncertainty is clearly communicated.

2.4. Application to selected stocks

As an example, we have summarized previous results from calculations of total mortality for a number of species and stocks in the large lakes. The methods and data sets used are summarized in Table 1. As expected, typical prey fishes generally have a higher total mortality than predatory fish. However, important commercial and predatory species like pikeperch also have higher total mortality than most other stocks, indicating that fisheries mortality most likely is more significant for this species (Figure 1).

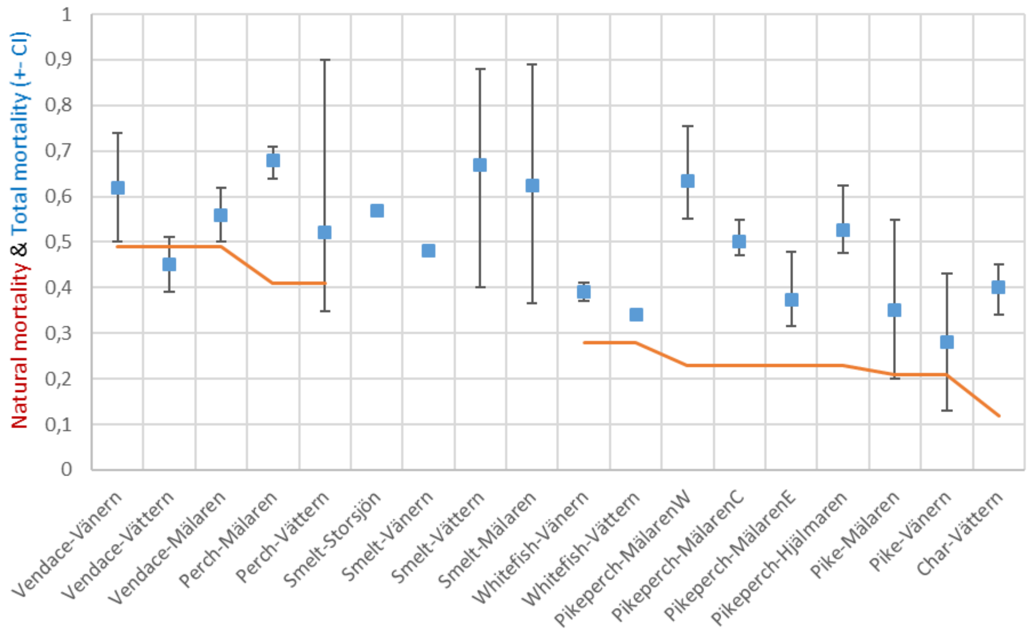


Figure 1. Total mortality estimated for a number of Swedish lake stocks (light blue squares) (± 95% confidence interval). Natural mortality (from the life-history tool at fishbase.se) (red/orange line) is also given as a comparison for all species besides smelt where such estimates were deemed not reliable.

Table 1. Methods and data sources of mortality analyses on lake fish stocks.

Species/lake/region	Years	Method/approach	Data from
Vendace-Vänern	**2010-2020**	**Chapman-Robson**	**Midwater trawling**
Vendace-Vättern	**2012-2016**	**Chapman-Robson**	**Midwater trawling**
Vendace-Mälaren	**2012-2016**	**Chapman-Robson**	**Midwater trawling**
Perch-Mälaren	**2021**	**Millar**	**Gill-net survey**
Perch-Vättern	**2017**	**Millar**	**Gill-net survey**
Smelt-Storsjön	**2011**	**Chapman-Robson**	**Midwater trawling**
Smelt-Vänern	**2019**	**Chapman-Robson**	**Midwater trawling**
Smelt-Vättern	**2019**	**Chapman-Robson**	**Midwater trawling**
Smelt-Mälaren	**2019**	**Chapman-Robson**	**Midwater trawling**
Whitefish-Vänern	**2020**	**Chapman-Robson**	**Gill-net survey**
Whitefish-Vättern	**2008-2015**	**Chapman-Robson**	**Gill-net survey**
Pikeperch-MälarenW	**2020**	**Millar**	**Trap nets**
Pikeperch-MälarenC	**2020**	**Millar**	**Trap nets**
Pikeperch-MälarenE	**2020**	**Millar**	**Trap nets**
Pikeperch-Hjälmaren	**2020**	**Millar**	**Trap nets**
Pike-Mälaren	**2015**	**Chapman-Robson**	**Trap nets**
Pike-Vänern	**2014**	**Chapman-Robson**	**Trap nets**
Arctic char-Vättern	**2005-2020**	**Millar**	**Gill-net survey**

In addition, we have compared total mortality of whitefish for L. Vänern and L. Vättern with four smaller lakes with a low fishing effort (Figure 2). Estimates of mortality were quite variable among lakes but in general slightly lower in L. Vättern and L. Vänern, indicating that fisheries as well as natural mortality in these lakes were relatively low prior to 2012 (Figure 3). This is not according to expectation; we interpret this because of the variation in natural mortality overriding moderate differences in fisheries mortality. The latest estimate of annual mortality in L. Vättern was around 30% per year (or Z=0.34) indicating that both fishing and predation related mortality are relatively low in this lake.

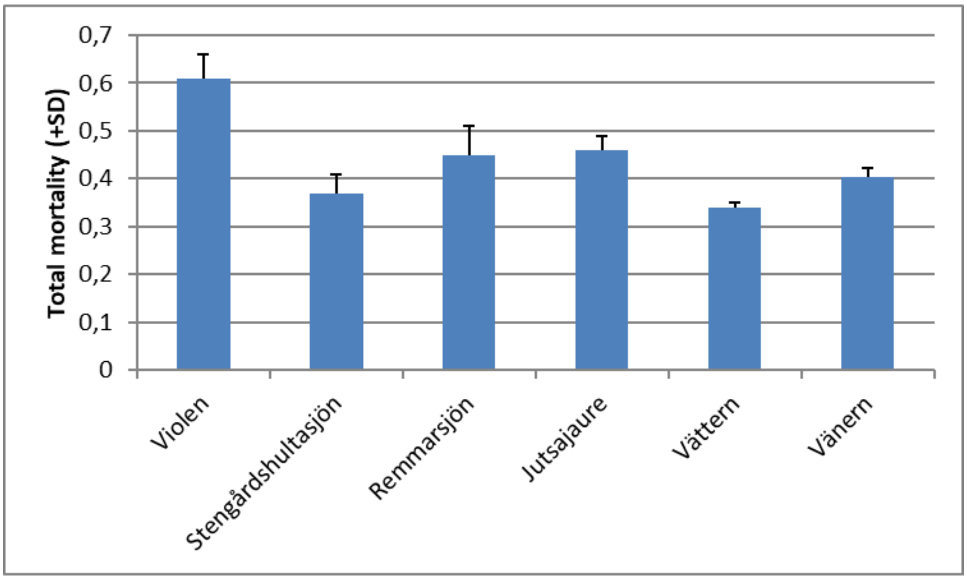


Figure 2. Comparison of total mortality (z ± confidence interval) of adult whitefish in various Swedish lakes using all available age/length data from the years 1996-2012. Data are from monitoring programs using multi-mesh gillnets. Catch data have been corrected for gear-selectivity using a bi-normal selectivity model. Mortality was assessed using the approach of Robson-Chapman (1960).

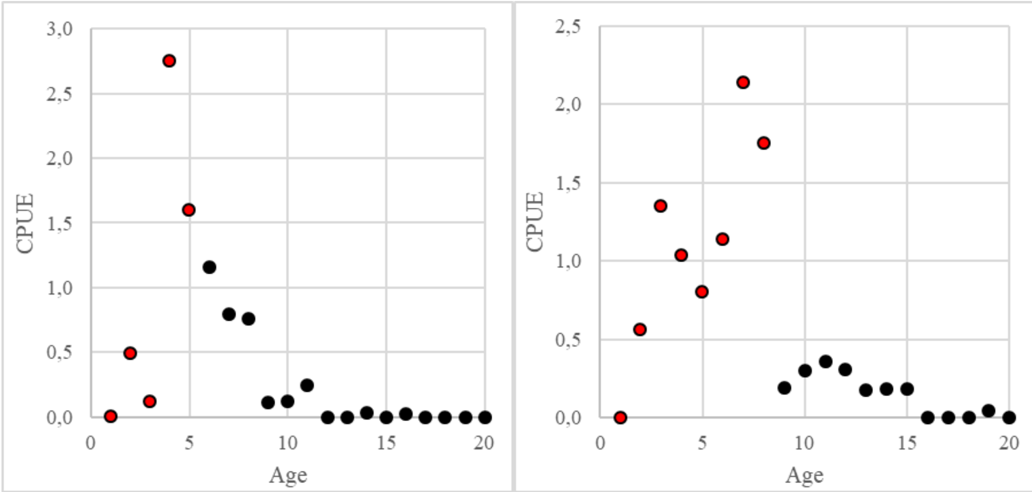


Figure 3. Catch in gillnet surveys of whitefish per age group in L. Vänern before (in 2012 - left) and three years after (in 2015 - right) that fishing seized. Red year-classes have not been subject to fishing and black year-classes experienced fishing. Note that mortality is calculated as the slope from the peak abundance, thus the shift from a period with fishing to a period without fishing enhances the slope.

The analyses (above) on mortality of various lake stocks indicate that total mortality is generally higher in small pelagic species compared to larger piscivorous and/or benthic species. Total mortality was also higher for all intensively exploited stocks, i.e. vendace in L. Vänern, pikeperch in western L. Mälaren and L. Hjälmaren and one of the perch stocks (Lake Mälaren). In addition, total mortality in relation to natural mortality was high in Arctic char in L. Vättern, another species for which the exploitation level most likely is relatively high. Overall, the results and general patterns appear as credible given what we know about the different stocks and how they are exploited. The confidence interval differed markedly between different stocks. This is related to several things: first, stocks with fishes that become old (L. Vättern whitefish as one example) have a lower confidence interval because the analyses follow a certain cohort over a long period of time; second, the confidence interval is smaller for stocks with a high number of age-read individuals. In some cases, there could also be differences in mortality between the localities covered in monitoring due to environmental differences, stock structure or fishing pressure that leads to a higher variability in mortality when data for the whole lake is pooled. One example is pikeperch in L. Mälaren where mortality analyses have to be performed for separate main basins due to marked difference in mortality. It is important to stress that the presented results on the different stocks should not be confused with SLUs annual assessment of stock status (HaV 2022). For some of these stocks (L. Mälaren and L. Hjälmaren pikeperch, L. Vänern vendace), quantitative stock assessments are being developed and will provide alternative measures of mortality (including fisheries mortality). The annual assessment of

Swedish national stocks (HaV 2022) also takes several other factors into consideration; thus they are not based solely on mortality. Total mortality estimates should generally also be used with care. As discussed previously there are several assumptions that need to be fulfilled for results to be unbiased and reliable.

2.5. Challenges and lessons learned

One important aspect to consider is how and when to correct size/age distributions due to gear selectivity (as recommended by (Thorson & Prager 2011). Gear selectivity has been modeled for some of the lake stocks and the best master curves were used to correct data and achieve less biased age/size data (Jonsson et al. 2013).

Another important and difficult part of performing a catch-curve analysis is how to assess peak age for a population when using a certain gear. Peak age is the age when a population is fully recruited to the specific gear. The catch-curve methods described above vary to some extent regarding at what age to start the analysis. Some start exactly at peak age and others at peak age plus one or two years (Ricker 1975). We have mainly used the approach recommended in Smith et al (2012). We modeled the peak abundance by fitting bell-shaped curves to the age vs gear selectivity corrected catch per unit of effort data. The best fit was obtained with a Weibull function. Peak age varied over time for some of the stocks. The peak age of Arctic charr for example increased by almost one year during the study period. We believe this was mainly due to slower growth of older fish in recent years leading to a shift in peak age.

The catch-curve analysis can be performed either on a specific cohort, following it over time in the catches or it can be performed on one year's catch, thus analysing several different cohorts. The cohort-based method was considered the preferable alternative, although it does require longer time series of data.

Many of the methods require specific assumptions to be fulfilled. These, however, are rarely met in their entirety in exploited populations. Thus, this analysis should be made with care and results should always be critically evaluated. One of the requirements of the catch curve method is that the fisheries and the age structure should be reasonably stabile and it is thus questionable to use this method during transition periods when conditions in the fisheries change, such as the years directly after the introductions of new fishing regulations. This shortcoming became apparent when performing a catch-curve analysis in L. Vänern where fishing for whitefish stopped in late 2011 (due to a lack of an exception for high dioxin levels). The year-classes (mainly those 4-8 years old) that had not been fished during 2012-

2015 were dominating the catch in monitoring in the year 2015 (Figure 3). This phenomenon pushed the peak age forward (from 4.9 to 6.9 years) and as a result of differences in catch between these cohorts and earlier ones that have been affected by fisheries the estimated total mortality appeared to increase after the ban. This is a rather illustrative example of a situation where results from these analyses should be carefully and critically evaluated, and where catch curves should preferably be avoided as a method for assessing total mortality until the population age structure has stabilized.

Some of the methods discussed above are hard to apply on crustaceans due to difficulties in performing accurate age analysis (see also the chapter on length-based indicators).

Another challenge with mortality estimates is that they normally require a relatively substantial sample of fish length and age. Given that the analysis focusses on following a specific cohort over time there is also a need for many years of substantial age and length data which is not available for some of the very most data-poor species and stocks (Coggins Jr. et al. 2013). There are several nationally managed stocks that are to data-poor for this analysis to be relevant. Some examples of such stocks that we worked with in this project is Baltic coastal whitefish stocks, Baltic coastal pikeperch stocks and smelt in L. Hjälmaren.

Mortality estimates can be performed on fisheries dependent as well as fisheries independent data. Data from monitoring has many advantages compared to data from fisheries but given that there is sufficient information about gear selectivity and that the sampling of the fishery has a sound design the method can be useful also for fisheries dependent data. Fisheries data usually has one advantage compared to monitoring data, the catch consists of a larger percentage of large and mature fishes (the right side of the peak age) which are the focal part of the population for catch-curve analysis.

In some lakes and many coastal areas there is more than one specific population occurring (whitefish is one relevant example). Different ecotypes can have different mortality patterns and should ideally be analyzed individually, but this is seldom possible since they are not accurately separated in catches. This is important to consider when utilizing results from mortality analyses on such mixed-population stocks. Genetic analyses as well as mark-recapture studies and demographic studies indicate that the population structure for many of the nationally managed fish species are complex (Östman et al. 2016; Andersson et al., 2015). Both in lakes and coastal areas there are examples of studies showing populations with more or less obvious isolation-by-distance patterns (Olsson et al. 2011; Wennerström et al., 2017) as well as studies that indicate the presence of more discrete stocks

(Dannewitz et al. 2011; Diaz-Suarez et al. 2022). For the majority of the nationally managed fish stocks, however, there is a lack of detailed knowledge of population structure. Thus, it is often difficult to provide stock-specific analyses of mortality (and other indicators) but it is important to address this issue when interpreting stock status and when formulating quantitative management objectives.

2.6. Recommendations

If possible, it is recommended that future analyses of mortality in exploited fish stocks can be conducted using more sophisticated analytical assessments. However, since this will not be possible for numerous stocks due to data and resource limitations, catch-curves and other simpler assessment methods could still be valuable. As mentioned above, mortality estimates should be carefully performed using best available methods and critically evaluated and compared with other biological information. Due to the limitations of these methods it could be valuable to consider pooling years to have more stable and reliable results.

Mark-recapture models and tagging studies using aquatic telemetry can provide an alternative way to assess mortality of stocks. Such studies are encouraged and hopefully they can provide valuable alternative data on mortality.

Mortality estimates are depending on collection of robust, long-term data on size and age of exploited stocks. It must be emphasized that such programs receive sufficient funding and that the design of data collection is sound. Several studies indicate that the number of individuals used in mortality analyses is important and that sample size needs to be at least 300-500 individuals per year out of which a substantial proportion needs to be analyzed for age (Coggins Jr. et al., 2013; Bohman et al., unpublished). At least 10 individuals per size bin, which in most cases means a minimum of 200 per year.

Lastly, there is abundant historic data sets on age and size structure of fishes and large archives containing biological tissues used for ageing. Assessment of mortality using historic data-sets could provide useful and important information, facilitating a better understanding of the population dynamics of data-poor national fish stocks.

3. Abundance/biomass related indicators

The most common indicator within fisheries science and management is the catch per unit of effort (CPUE) index. It can be expressed as either an abundance indicator, by measuring the number of individuals per unit effort, or as a biomass indicator, when calculated as weight per unit of effort (WPUE). The data can come from either a survey or from fisheries, although most commonly from the commercial rather than the recreational fisheries data (Scandol 2005). The basic concept is that changes in CPUE reflect changes in fish abundance as the effect of differences in effort to catch them has been removed. The indicator thus rests on the assumption that CPUE is proportional to abundance and that catchability of the stock is constant over time. However, several factors can influence catchability and information about environmental parameters, size composition, recruitment and fishing effort can provide help to interpret potential trends in abundance/biomass of stock (Heessen & Daan 1996; Naddafi et al. 2022).

In some cases, only commercial catch data, without effort, is available. Interpreting catch-only data can be difficult as low catches can be indicative of both under- and overexploitation and high catches can reflect both unsustainability in the stock (overexploitation) or reflect a high recruitment in a specific time period. Nevertheless, there are also methods for providing catch advice based on reliable catch-only data (ICES 2012). However, catch-only methods are generally very uncertain and risk producing biased estimates of stock biomass (Free et al. 2020; Ovando et al. 2022). But, the use of ensemble methods, where multiple models are used together can provide useful information, particularly if they are interpreted with care. A nice recent example is a German study on coastal pike, where an ensemble approach utilizing both commercial and recreational statistics was used (van Gemert et al. 2022).

Information about population dynamics and fisheries in combination with fishing effort data and commercial catch of the target species can help to interpret trends in catch data and exploitation status of the target stock in data-poor conditions (Vasconcellos et al. 2005). In fact, there are several assessment methods that rely on the combination of catch data and CPUE indices, from either survey or catch data, which are combined in various modelling frameworks, e.g. surplus production models (SPMs) (ICES 2012, 2022).

3.1. Formulation of objectives and targets

”Maintain population sizes above half of natural abundance”

One of the rules that minimize the impact of fishing on commercial species is to maintain population sizes above half of natural abundance, at levels where populations are still likely to be able to fulfil their ecosystem functions as prey or predator (Froese et al. 2016). This rule can provide help to achieve several goals of ecosystem-based fisheries management. For instance, the biomass of prey and predator species can be rebuilt in the system and thus the impact of fishing is diminished (Froese et al. 2016).

“Populations should be above “Bmsy”

BMSY is the average long-term stock biomass corresponding to a harvest of maximum sustainable yield (MSY) and a fisheries mortality of Fmsy. Harvest rates approaching MSY will maximize the natural compensatory mechanisms of the exploited stock. Faster growth, higher fecundity, lower levels of cannibalism etc will thus compensate depletion of stock biomass due to fishing. According to EU and SwAM, the population target should be to keep stock size larger than BMSY. To achieve this target, the fishing mortality threshold (the threshold above which a stock is experiencing overfishing) should be smaller than the fishing mortality that would produce BMSY (i.e. FMSY) (Cooper 2006). For many fish stocks, fishing mortality should be less than natural mortality rate, which is indeed an upper limit for FMSY (Cooper 2006).

“At least 40 % of the adult spawning potential (roughly - 40 % of SSB)” (Hordyk et al., 2015)”

The spawning potential ratio (SPR), the proportion of the unfished reproductive potential left at any given level of fishing pressure (Walters & Martell 2004), is used to restrict reference points and set targets for fisheries (Hordyk et al. 2015). A fishing mortality threshold is often set to F40% (Cooper 2006). F40% is the fishing mortality rate that results in SPR = 40%, which is considered risk adverse for many species (Clark 2002). In other words, this fishing mortality should decrease the spawning stock biomass per recruit to 40 percent of what would exist in the absence of fishing (Cooper 2006). Length-based spawning potential ratio (LB-SPR) is described further in the chapter on size- and age-based indicators (Hordyk et al. 2015).

3.2. How do the indicators work?

The following abundance indicators were used in this report:

3.2.1. Status Quo index (SQ)

The Status Quo index is used within ICES as a "harvest control rule" that combines a survey abundance/biomass index with catch or landings data to provide a catch advice for exploited stocks (ICES 2012). The general concept is based on comparing a recent period relative to the preceding years and if the survey index indicates an increase, the suggested catch for next year can be incrementally increased as well, or the opposite if the survey index is decreasing. The recent period can either refer to the most recent year, or an average over 3 years or longer for long-lived organisms, and it is recommended that the number of years included should account for interannual variability of the surveys (ICES 2012). In general, ICES suggest using the average BPUE (Biomass Per Unit of Effort) of the last two years (A, e.g. 2020, 2019) divided by the average of the prior three years (B, 2018, 2017, 2016) (defined as short term periods (5 years) SQ index in this report), hence: A/B = index ratio for population density. If the goal is to maintain the status quo (i.e. an index ratio = 1), future catches are regulated as a proportion to the ratio from the survey. For example, if A/B > 1, the index is increasing, and, hence, catches can increase in proportion to A/B. On the contrary, if A/B < 1, the population density is decreasing, catches should be reduced in proportion to A/B. Furthermore, if the index is negative, a precautionary buffer (a cap) at 20% fishery reduction above the proportional calculation is recommended.

3.2.2. ASCETS

Analyses of structural changes in ecological time series (ASCETS) has been suggested as a generic quantitative tool; applicable to a wide range of ecological time-series for assessing changes in indicator state (Östman et al. 2020). It can be used to assess changes in the status of coastal fish communities (Naddafi et al., 2022; Helcom 2023a). Other fishery models such as Status-quo harvest control and Depletion-Corrected Average Catch (ICES 2012) require catch data that is normally either not available (due to no fisheries data collection) or highly uncertain (e.g. from recreational fisheries) for most nationally managed fish populations. Further, these methods are sensitive to both single observation errors and environmental stochasticity. In contrast, ASCETS focuses on longer time-series (preferably >10 observations) and encompasses an array of natural and human induced drivers as well as observational errors, and only assumes that random processes are identical across the reference period (Östman et al. 2020). To determine the status of the indicator, the ASCETS method first derives a bootstrapped distribution of median values from a time series of observed indicator values during a reference period (Helcom 2023a). Specific threshold values for changes in indicator state is set, and for each species, these are based on the 5th and 98th percentile values of the bootstrapped distribution (Helcom 2023a). In this way, the derived boundaries of this interval can function as threshold values for a change in state per assessment unit of each species (Helcom 2023a). Second, the bootstrapped median indicator value during the assessment period is evaluated in relation to the threshold values derived from the reference period depending on how much of the bootstrapped median distribution from the assessment period that falls below, within, or above the 5th and 98th percentiles (Helcom 2023a).

3.2.3. Surplus production models

Surplus production models (SPMs) are widely used in fisheries management and as the name suggests they rely on the theory of maintaining a population below its carrying capacity, which is the maximum upper limit to the size of a stock, and fishing on the surplus production (Cousido-Rocha et al. 2022). When reliable information on natural mortality, length and age structure are not available, surplus production models can be used to estimate exploitation and stock status of a species (Beverton & Holt 1957). In such models, length- and age structure, and other characteristics such as sex-specific growth, of the target species is not considered, and sustainable exploitation is assumed to be the function of one large unit of population biomass (Holt 2014; Cousido-Rocha et al. 2022). Surplus production models, developed by Fox Jr. (1970) and Schaefer (1954), were originally based on differential equations and an equilibrium assumption describing a stock at its carrying capacity. In the Fox (1970) model, maximum sustainable yield (MSY) is obtained at 37% of carrying capacity, and in the Schaefer (1954) model at 50% of carrying capacity. The equilibrium assumption however leads to over-estimated catch recommendations and has caused collapse of several stocks, including the Peruvian anchovy in the 1970s (see references in (Cousido-Rocha et al. 2022). However, as methodologies that do not require equilibrium assumptions for model fitting, the assessment method has seen a renewed interest since the 1990s (Cousido-Rocha et al. 2022). There are several different SPMs available, but perhaps the two most common methods include the “surplus production in continuous time” - SPiCT (Pedersen & Berg 2017) and the “just another Bayesian biomass assessment” – JABBA (Winker et al. 2018), and its extension JABBA

SELECT (Winker et al. 2020). Which method to choose depends on the application, but in short, SPiCT is a slightly more complex model and does not rely on the very strong assumption of a linear relationship between the catch rate and stock biomass that JABBA does (Cousido-Rocha et al. 2022).

In general, the minimum data requirements are i) an index of relative exploitable biomass and ii) a time series of associated catch data. The index can be either fisheries dependent or independent, i.e. either commercial or survey catch-per-unit-effort. As the models rely heavily on the index, it is good practice to first "standardize" the index by taking other influential factors into account (Maunder & Punt 2004). For example, in the following SPM (3.3.3), CPUE data was first standardized using generalised linear models (GLM) to consider the effects of year, temperature, and depth on the data and then used in JABBA model. Importantly, the biomass index, especially if derived from survey data, must only include the "exploitable stock biomass" (i.e. sizes that are represented in the commercial catches, e.g. González Herraiz et al. 2023) in order to represent the vulnerability to fishing. It is also beneficial for the fit of the model if the times series covers a larger range of values, i.e. periods of both low and high catch and index, in order to find contrasts in the data (Cousido-Rocha et al. 2022). Insufficient contrast can lead to estimation problems, especially for models that start after the stock has been heavily exploited (Hilborn 1979). Lastly, an additional recommendation is that the catch data used in the model is representative of total removals (landings, discards and by-catch as well as recreational catches) where relevant and possible (Jakubavičiūtė et al. 2022).

3.3. Application to selected stocks

3.3.1. Status Quo index (SQ)

Although quantitative catch advice, i.e. quotas, is not requested by fisheries managers for national stocks, we illustrate the applicability of the SQ-approach using survey data. The survey data come from coastal monitoring using standardized gillnets. We calculated a biomass index (biomass per unit effort – BPUE) as total biomass (in kg) for each species per local and number of gillnets (i.e. weight per net and night). In total, we calculated the SQ for pikeperch in three sites (Galtfjärden, Kvädöfjärden, and Forsmark), pike in three sites (Asköfjärden, Torhamn, and Kvädöfjärden), and whitefish in four sites (Holmöarna, Norrbyn, Kinnbäcksfjärden, and Kvädöfjärden).

Here, short term periods (5 years) SQ index were excluded because the number of fish caught by gill nets for our three data poor stocks was relatively low during the last five years. Instead, by using long-term period data (15 years), we included more BPUE data and took more years into account; an approach that may provide more realistic results. In order to estimate SQ index based on long-term data, we used the average BPUE of the last five years (A, 2016–2020) divided by the average of the 10 prior years (B, 2006–2015), hence: A/B = index ratio.

Pikeperch

Our survey data revealed a low abundance of pikeperch in all three monitoring areas over time (Figure 4). The SQ index was smaller than 1 in both Galtfjärden and Forsmark, indicating that the population density of pikeperch has decreased in these two coastal areas (Figure 5).

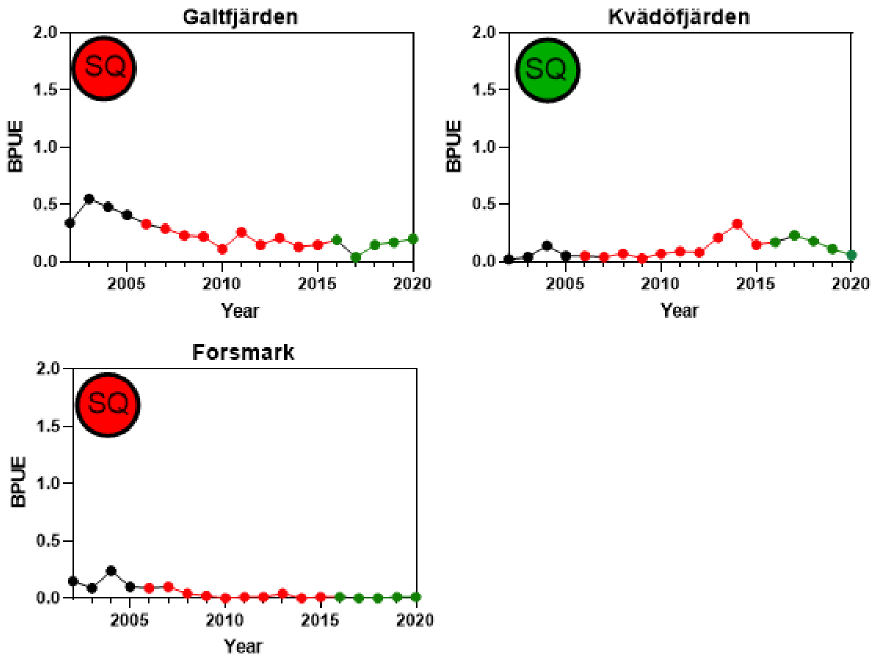


Figure 4. BPUE (kg) of pikeperch in three monitoring areas. Red Status Quo (SQ) symbol indicates an index ratio (A/B) <1, green SQ-symbol indicates an index ratio (A/B) >1. Green dots on the lines show "A" (the average BPUE of the recent years) and red dots indicate "B" (the average BPUE of the previous years).

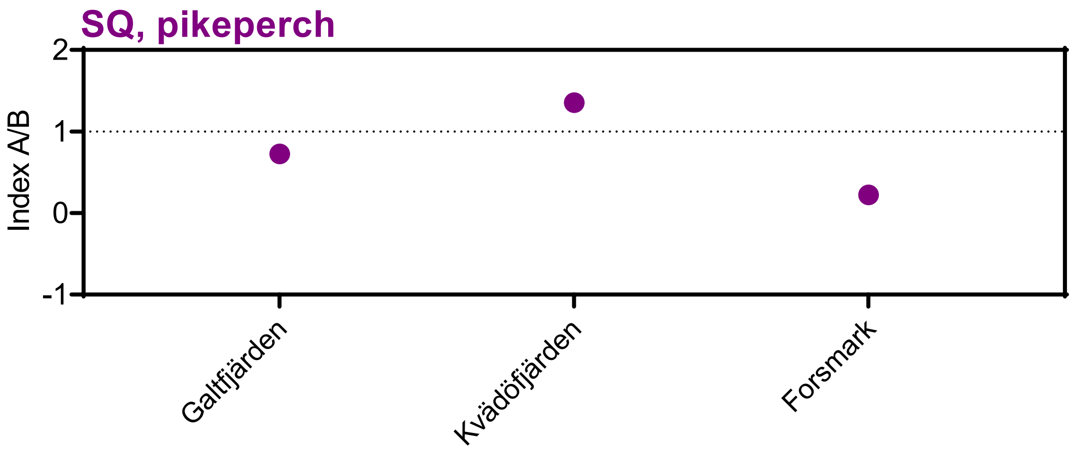
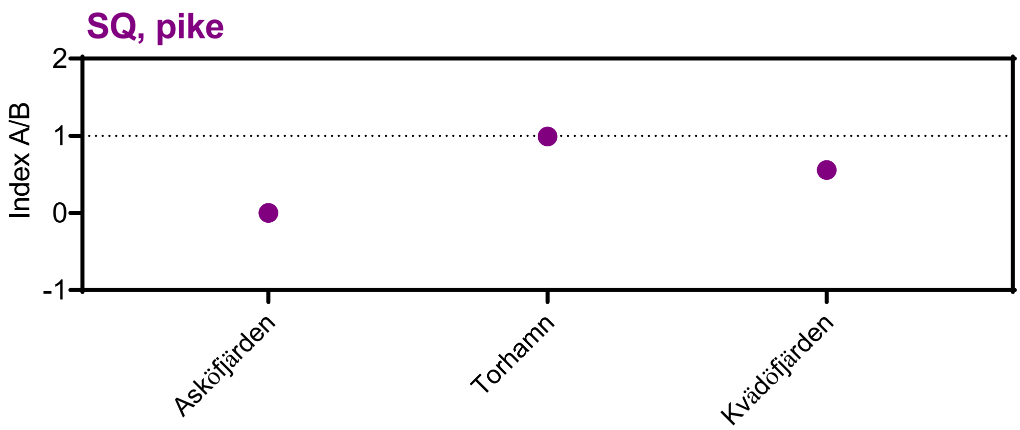


Figure 5. Estimated SQ values (index A/B) for pikeperch in three monitoring sites. Dashed line shows the threshold of 1 for SQ index. "A" is the average BPUE of the recent years (2016-2020) and "B" is the average BPUE of the previous years (2006-2015).

Pike

A small number of pike were caught by multi mesh size gillnets in all three monitoring areas (Figure 6). In general, the SQ index was smaller than 1, indicating that the population density of pike has decreased in these coastal areas (Figure 7). However, gillnets may not be the best gear to catch pike since pike are stationary and have very low catchability in gillnets.



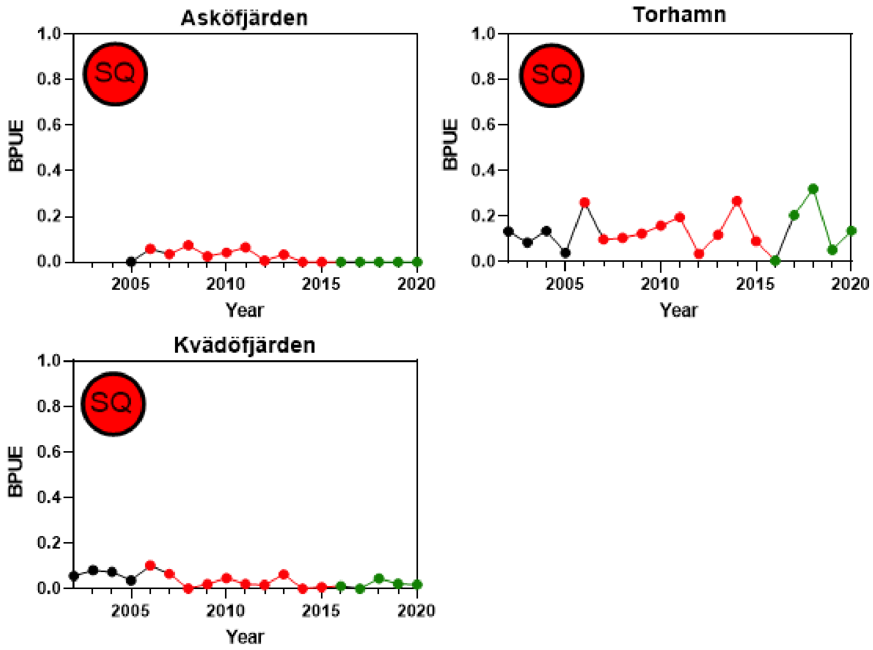


Figure 6. BPUE (kg) of pike in three monitoring areas. Red Status Quo (SQ) symbol indicates an index ratio <1. Refer to Figure 4, for red and green dots on the lines.

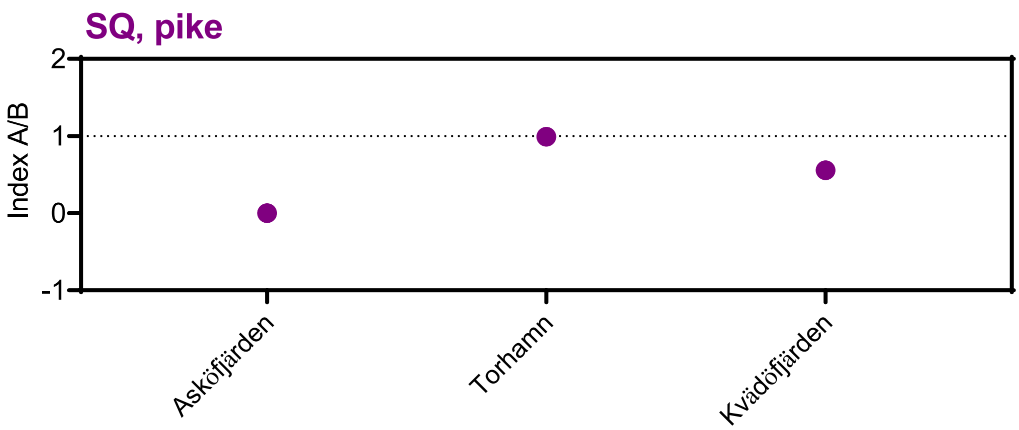


Figure 7. Estimated SQ values (Index A/B) for pike in three monitoring sites. Dashed line shows the threshold of 1 for SQ index. Refer to Figure 5, for A and B.

Whitefish

BPUE (kg) of whitefish varied among monitoring sites and years (Figure 8). SQ index values were larger than 1 in half of the monitoring sites (Holmöarna, Norrbyn, and Kinnbäcksfjärden) indicating an increased whitefish population density in these areas (Figure 8 & Figure 9). This positive trend is probably due to fisheries regulation implemented about 10 years ago in these areas that has not allowed gillnet fisheries shallower than 3 m in large part of the year. In Lagnö, the SQ index shows that the whitefish population density has increased (Figure 8 & Figure 9).

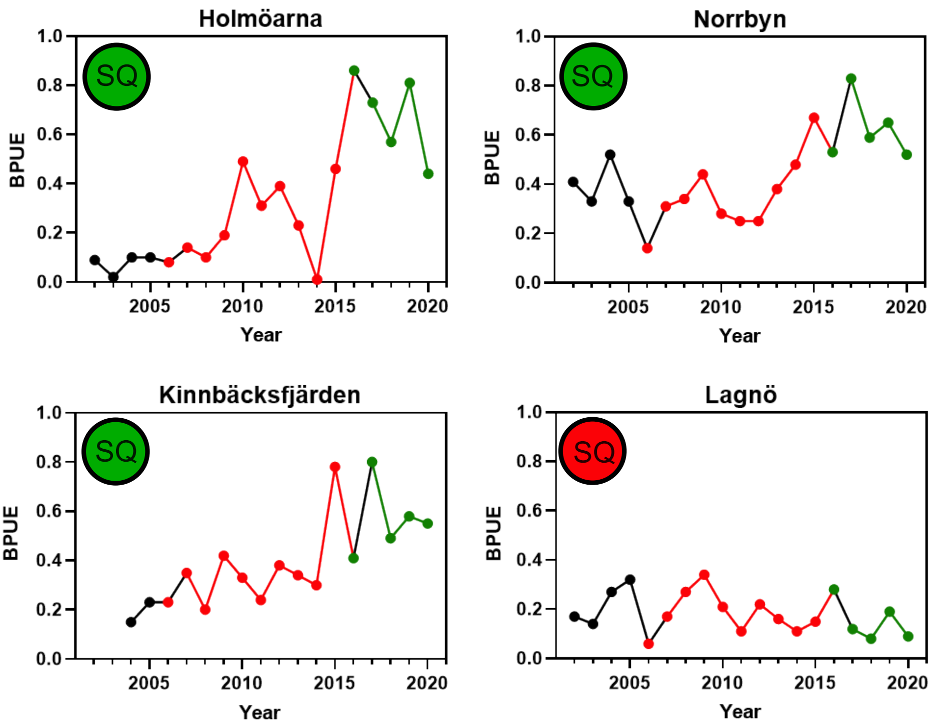


Figure 8. BPUE (kg) of whitefish in different monitoring areas. Red Status Quo (SQ) symbol indicates an index ratio <1, green SQ-symbol indicates an index ratio >1. Refer to Figure 4, for red and green dots on the lines.

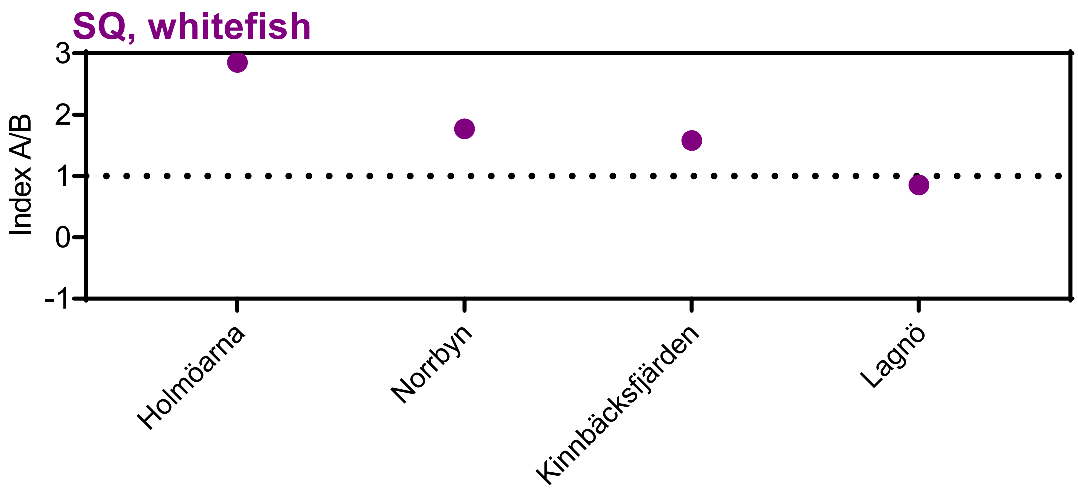
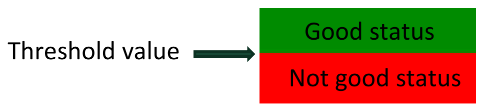


Figure 9. Estimated SQ values (index A/B) for whitefish in four monitoring sites. Dashed line shows the threshold of 1 for SQ index. Refer to Figure 5, for A and B.

3.3.2. ASCETS

Good Status for a species is achieved when its abundance is above a specified threshold value. ASCETS establish threshold values for the state during a reference period based on the observed variation in indicator values. Here we use indicator values during a pre-defined reference period (2002-2015 in Forsmark, Holmöarna, Kvädöfjärden, Lagnö, Norrbyn, and Torhamn; 2004–2015 in Gaviksfjärden and Kinnbäcksfjärden); 2005–2015 in Asköfjärden) relative to median indicator values during the assessment period of the last five years in each data set (2016-2020). In ASCETS, good status is evaluated based on the deviation of the median value of the indicator during the assessment period in relation to the threshold value as follows:



Pikeperch

Good status for pikeperch was achieved only in Kvädöfjärden whereas the status for pikeperch was poor in Galtfjärden and Forsmark (Figure 10).

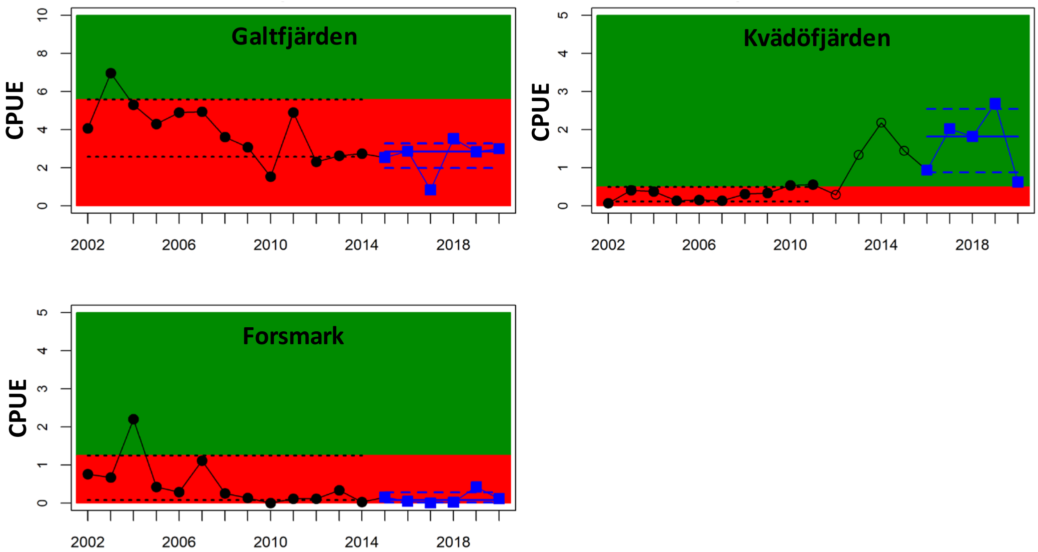


Figure 10. Status evaluation of pikeperch for three coastal areas. Threshold values representing the 5th and 98th percentiles of the resampled median values during the reference period are shown by black dotted lines between fields in green (good status) and red (not good status), with the colour of the fields determined by the status during the reference period. The evaluation of good status/not good status is done for the assessment period compared to the reference period by comparing the location of the median during the assessment period (full blue line) with the location of the threshold lines. The 95th percentile intervals associated with the median of the assessment period are shown in hatched blue lines. The confidence in a change of status between the reference and the assessment period is determined by how much of the bootstrapped median distribution from the assessment period that falls below, within, or above the 5th and 98th percentiles of the reference period.

Pike

The status for pike was poor in all three studied coastal areas (Figure 11).

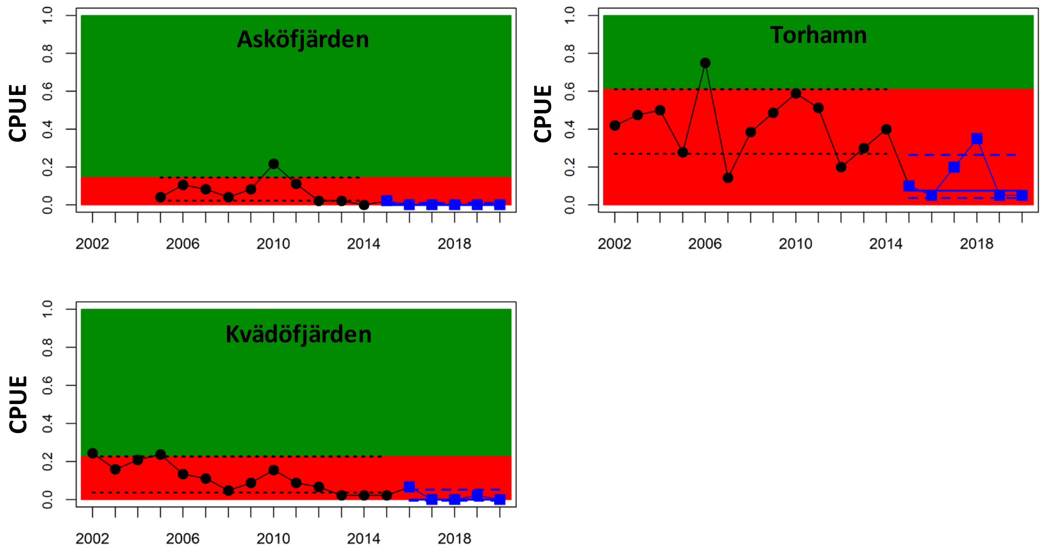


Figure 11. Status evaluation of pike for three coastal areas. Threshold values representing the 5th and 98th percentiles of the resampled median values during the reference period are shown by black dotted lines between fields in green (good status) and red (not good status), with the colour of the fields determined by the status during the reference period. The evaluation of good status/not good status is done for the assessment period compared to the reference period by comparing the location of the median during the assessment period (full blue line) with the location of the threshold lines. The 95th percentile intervals associated with the median of the assessment period are shown in hatched blue lines. The confidence in a change of status between the reference and the assessment period is determined by how much of the bootstrapped median distribution from the assessment period that falls below, within, or above the 5th and 98th percentiles of the reference period.

Whitefish

Whitefish did not reach the threshold for good status in Lagnö (Figure 12). The abundance of whitefish in CPUE indicates that the stock reaches good status in Holmöarna, Norrbyn, and Kinnbäcksfjärden (Figure 12).

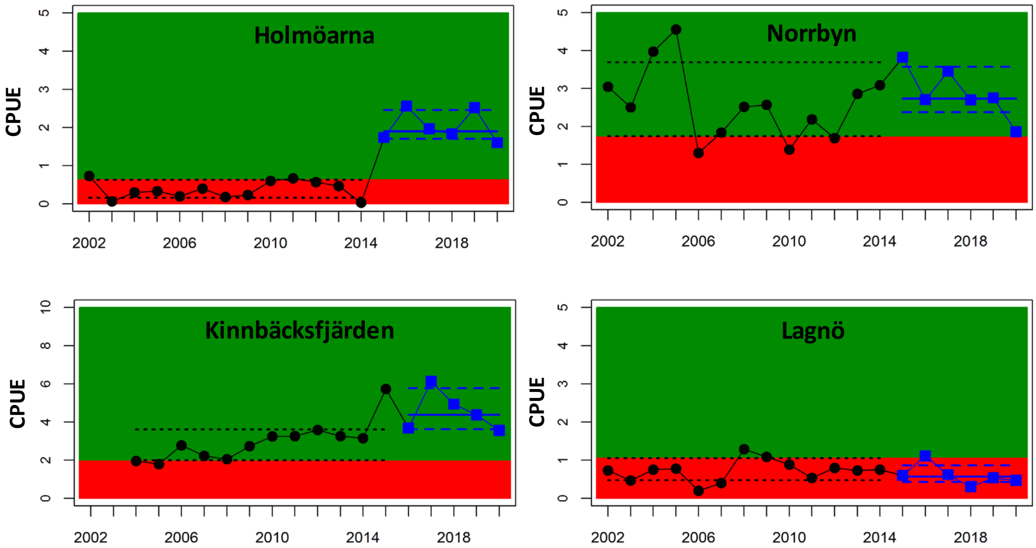


Figure 12. Status evaluation of whitefish for four coastal areas. Threshold values representing the 5th and 98th percentiles of the resampled median values during the reference period are shown by black dotted lines between fields in green (good status) and red (not good status), with the colour of the fields determined by the status during the reference period. The evaluation of good status/not good status is done for the assessment period compared to the reference period by comparing the location of the median during the assessment period (full blue line) with the location of the threshold lines. The 95th percentile intervals associated with the median of the assessment period are shown in hatched blue lines. The confidence in a change of status between the reference and the assessment period is determined by how much of the bootstrapped median distribution from the assessment period that falls below, within, or above the 5th and 98th percentiles of the reference period.

3.3.3. Surplus production models

The open-source Bayesian State-Space Surplus Production Model framework, JABBA (Winker et al. 2018) was applied for the pikeperch stocks in Galtfjärden and Kvädöfjärden (Jakubavičiūtė et al. 2022). In this model, incorporation of both observational and process errors is enabled by the state-space framework and uncertainty in the parameter estimates is quantified by the Bayesian approach (Jakubavičiūtė et al. 2022). Here, following recommendations in ICES 2022, the Schaefer type production curve (Schaefer 1954), which assumes maximum sustainable biomass at 50% of carrying capacity, was used (Jakubavičiūtė et al. 2022). Independent fisheries data or survey CPUE as well as catch time series related to the pikeperch stocks in Galtfjärden and Kvädöfjärden were used in the JABBA model. In this model, the assumption is that the abundance index (CPUE) is an informative index of relative stock abundance. However, these indices included also smaller sizes than minimum allowable catch sizes of pikeperch and it can be discussed if the index therefore represent “exploitable biomass”. For details about this model, refer to Jakubavičiūtė et al. (2022) and see also the SQ-assessment above.

The result of the model shows that, in Galtfjärden, stock spawning biomass was below the biomass at maximum sustainable yield (Bmsy), indicating that the pikeperch stock is in a depleted state (Figure 13, see also Jakubavičiūtė et al. (2022)). In contrast, the stock spawning biomass has been higher than estimated Bmsy in Kvädöfjärden since 2010 indicating that the stock status has improved during the last decade (Figure 13). Indeed, a positive trend was observed in the Kvädöfjärden stock during 2010-2020. However, the Galtfjärden stock shows a declining population trend (Figure 13), revealing a concerning situation for this stock. The model proposed a relatively good status of the pikeperch population in Kvädöfjärden, which corresponds well with SQ and ASCETS results. However, the assessed time period is limited and does not include substantially larger landings observed in earlier decades. Thus, important data contrasts that could affect model fit was not accounted for during model fitting.

Swedish commercial catches in the Sea of Åland and Bothnian Sea, which comprises Galtfjärden, as well as in the Northern Baltic Proper, which includes Kvädöfjärden, have generally been decreasing from 1994 to 2020 (Figure 14). However, commercial catches were relatively high during 2005-2007 in the Sea of Åland and the Bothnian Sea (Figure 14). It should be noted that the current model did not include recreational catches nor predation from seals. Unfortunately, recreational catches are not available at that geographic resolution, but at the larger scale (basin-wide), recreational catches appear between 5 and 10 times larger than commercial catches, and the model therefore markedly underestimates total landings. This illustrates also a more general difficulty that can potentially affect most indicator and assessment frameworks. What happens with management goals and reference points if/when they are set based on incomplete data?

It should be noted that SPMs do not account for age- or size structure. To illustrate how both biomass and size is needed for a more holistic picture of stock status, we have calculated four length indicators for the pikeperch stocks (Figure 15, see the chapter on size- and age-based indicators for further details on how they are calculated). In Galtfjärden, the analyses indicate a declining trend in the largest sized fish, both for the indicator $L_{max5\%}$, i.e. the size of the largest 5% ($F_{(1,17)}$=14.05, estimate=-0.57 (se=0.15), $p<0.001$, R^2=0.43) and L90, the 90th percentile of the length distribution ($F_{(1,17)}$=13.46, estimate=-0.38 (se=0.10), $p<0.01$, R^2=0.44). There was also a small increase in the indicator for the size of the smallest individuals ($F_{(1,17)}$=8.87, estimate=0.07 (se=0.02), $p<0.01$, R^2=0.34), which we interpret as driven by the larger decrease in large fish (these indicators are co-dependent as they are derived from the same size distribution, in this case fish >15 cm). In Kvädöfjärden, the indicator for the smallest sizes, L10, is decreasing ($F_{(1,17)}$=13.2, estimate =-0.58 (se=0.16), p=0.002, R^2=0.44), as is the median size (indicator L50, $F_{(1,17)}$=7.76, estimate =-0.59 (se=0.21), p=0.013, R^2=0.31). The indicators for the largest sized fish show no statistically significant trend, for neither $L_{max5\%}$ ($F_{(1,17)}$=0.24, estimate=-0.19 (se=0.39), p=0.63, R^2=0.01) nor L90 ($F_{(1,17)}$=3.09, estimate=-0.36 (se=0.20), p=0.097, R^2=0.15). Taken together, the increased biomass index, as shown in the SPM for Kvädöfjärden (Figure 13), appears to be driven by an increase in smaller sized pikeperch, which are not (yet) exploitable. In general, the estimates based on independent fisheries survey, which uses standardized sampling methods, can be biased towards smaller individuals compared to commercial gears. However, low number of pikeperch caught in Kvädöfjärden, especially in the beginning of the time series, are too low for accurately describing the size structure of the stock (Figure 15). The numbers caught in Galtfjärden are slightly higher, but still too low to yield precise estimates of the length indicators, although they here appear useful for detecting long-term trends. Qualitatively, no pikeperch above the current minimum size (45 cm) has been caught in Galtfjärden since 2003, and in Kvädöfjärden only sporadic catches of 1-3 individuals occurred up until 2018 (HaV 2022), suggesting a poor state for coastal pikeperch stocks. The formal stock assessment from SLU for coastal pikeperch, which includes not only biomass but also size composition and mortality, is that catches should be reduced (HaV 2022) and that coastal pikeperch is “outside of biologically safe limits” in the latest assessment (Fiskbarometern, unpublished).

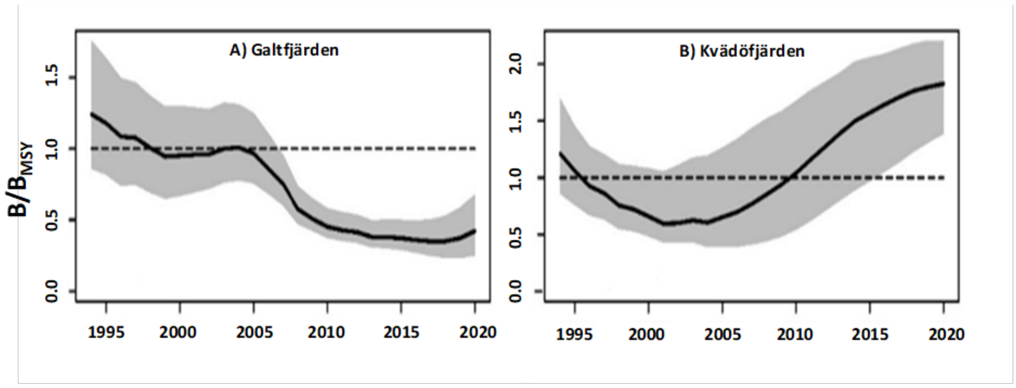


Figure 13. Predicted pikeperch biomass changes in Galtfjärden and Kvädöfjärden. The y axis shows biomass relative to biomass at maximum sustainable yield (Bmsy). The grey area shows the 95% credibility interval, and the thick central line shows the median of the posterior distribution of the model estimates.

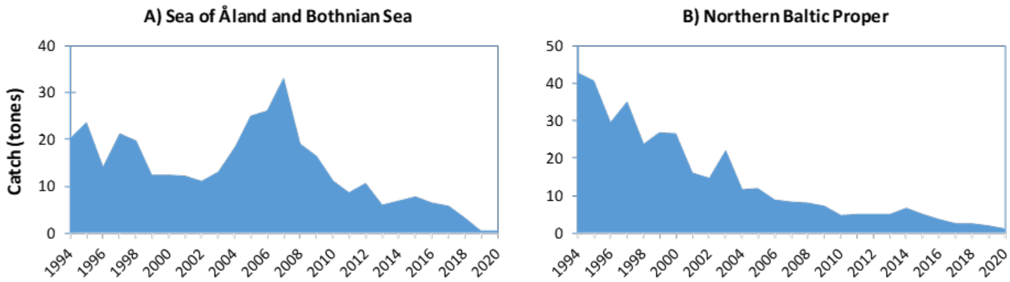


Figure 14. Swedish commercial catch of pikeperch in Sea of Åland and Bothnian Sea (A) and the Northern Baltic Proper (B).

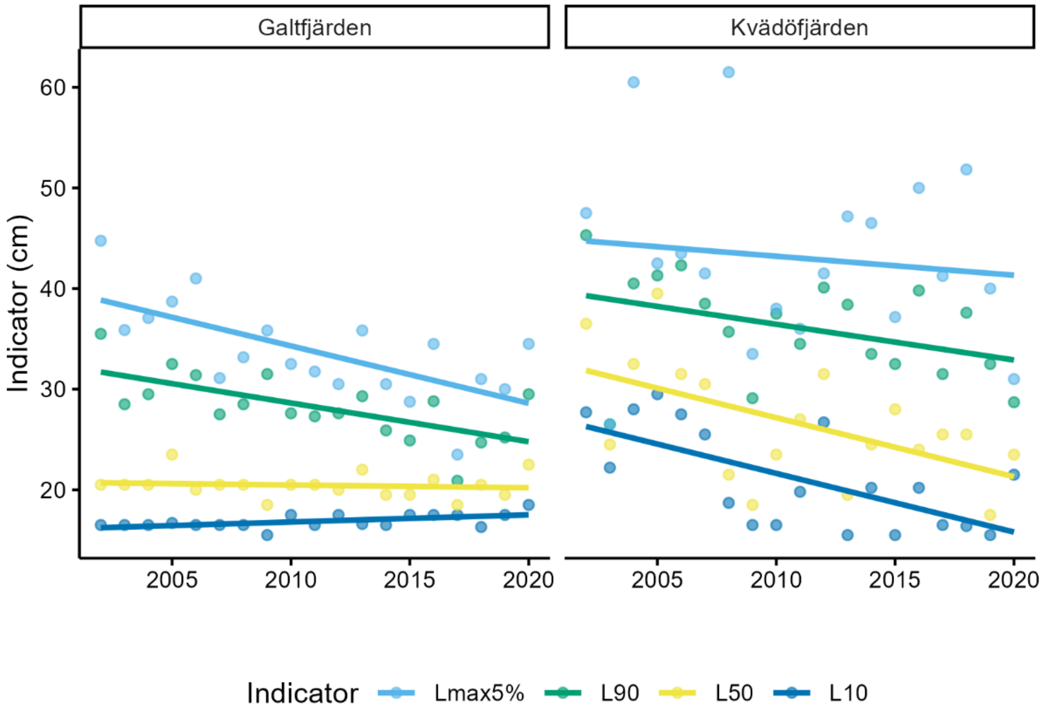


Figure 15. Four length-based indicators applied on pikeperch from standardized monitoring in Galtfjärden (left) and Kvädöfjärden (right). Only pikeperch >15 cm have been included in the analyses. The total number of pikeperch (>15 cm) caught per year was for Kvädöfjärden: 2, 18, 16, 5, 7, 6, 13, 15, 21,24, 13, 42, 88, 42, 38, 75, 60, 106, 29, and for Galtfjärden: 61, 151, 125, 83, 72, 90, 60, 59, 34, 73, 50, 52, 47, 63, 58, 17, 39, 64, 44, from 2002 to 2020, respectively. The indicators are described in the size- and age-chapter (Lmax5% is based on the average size of the 5% largest individuals per year).

3.4. Challenges and lessons learned

Both catch-based methods as well as surplus production assessments risk misclassification of stock status. Based on simulations, catch-based methods misclassified the status of about two-thirds of the stock, while SPMs performed slightly better with a misclassification in about 40% of the stocks (Carruthers et al.

2012), see also (Carruthers et al. 2014). Thus, the recommendation is to use these types of methods carefully and not solely rely on these types of assessment methods. Carruthers et al. (2014) also assessed the status quo method specifically and concluded that it performed poorly in cases where stocks are below their most productive levels.

Fisheries dependent CPUE data may not precisely reflect fish abundance if catchability varies over time. Such CPUE data are also highly variable as they depend on several other factors, e.g. gear selectivity, natural mortality, season, management measures, price and other factors (Hilborn & Walters 1992; Harley et al. 2001). To overcome these problems, it is recommended that fishery-dependent CPUE data should be standardized as regards methods and models (Maunder & Punt 2004; Carruthers et al. 2011).

In independent fisheries surveys, sampling is performed a specific time every year with standardized design and gears. Still, species catchability (and thus indicator values) may be affected by individual size and change in spatial distribution and movement of stocks (Hilborn & Walters 1992). However, the information about environmental parameters, size composition, recruitment and fishing effort can provide help to interpret potential trends in abundance/biomass of stock (Heessen & Daan 1996). In fact, abiotic and biotic environmental variables may limit the indicators' ability to provide confident results and prevent accurate assessments and thereby influence management action (Bergström et al. 2016a; b; Östman et al. 2017). For instance, coastal fish species, which are a resource for commercial and recreational fisheries and key ecosystem components in the Baltic Sea, is used as management objectives within the EU Marine Strategy Framework Directive and the HELCOM Baltic Sea Action Plan. Using 16 years of monitoring data, over a latitudinal range of 56 – 66°N along the Swedish Baltic Sea coast, Naddafi et al. (2022) evaluated the effect of variability in water temperature and depth, and wave exposure for three indicators of environmental status assessment in the Baltic Sea: Abundance of perch, Abundance of Cyprinids, and Abundance of Piscivores (Naddafi et al. 2022). Generalized linear mixed models (GLMM) revealed an overall positive linear relationship between water temperature for all indicators, and overall negative linear relationships to depth and wave exposure (Naddafi et al. 2022). When adjusting indicator values using the parameter estimates from the GLMM models, the variability and 95 % confidence interval for all three indicators were reduced (Naddafi et al. 2022). The results suggested that adjusting coastal fish indicators to variation in local ambient environmental factors will increase their precision, and hence, the confidence in the assessment of environmental status (Naddafi et al. 2022).

Another potential problem is related to the difficulty of calculating reference points, which needs abundance and virgin biomass (B_0) data. It is difficult to estimate absolute abundance and virgin biomass very reliably (Hilborn 2002). When stock-assessment scientists rely on only reference points, they may neglect projections and evaluation of alternative management policies and thus avoid addressing more significant problems in fisheries management (Hilborn 2002). Sometimes, inappropriate reference points are applied to species for which they were not derived (Hilborn 2002). Furthermore, high availability and quality of data are required for an assessment resulting in analytical reference points, such as B_{msy}, which is currently plausible only for a small set of species (Flather et al., 2011, Borja et al., 2013). Lack of important reference points limits the variety of available objectives that can be used in management.

As illustrated by pikeperch above, SPMs do not consider age- or size structure of the studied stocks. Thus, methods that rely on biomass and maximizing yield may indicate that the stock is sustainably fished, while at the same time there is a parallel loss of large and old fish that is not included in the assessment. However, also data-rich stocks with analytical assessments have experienced problems with incorporating effects of size-selective harvesting in the modelling framework (Cardinale & Hjelm 2012). This highlights the need of including both abundance and biomass-based indicators, as well as size- and age structure in the assessment frameworks.

3.5. Recommendations

- We conclude that it is important to try to use the best available methods and to carefully scrutinize assumptions to ensure that the data and method is compatible.

- Standardize CPUE indicators. Depending on data sources, different factors need to be accounted for, e.g. site, year, gear, and abiotic factors, such as temperature and depth.

- Do not rely solely on CPUE-indices, catch-based methods or surplus production models for assessing stock status – include also size- (like LIME model) or age composition of the stock. In addition, methods like SQ assess only trends in abundance without considering if the level of population density is within safe biological limits. Hence, such methods need to be carefully used and those trends should be assessed with some consideration of levels.

4. Size and age-based indicators

Given that fishing affects size structure of fish populations and accordingly life history traits like mortality, maturity, and reproductive output, size-based indictors can be valuable when assessing the status of a fish stock (Blanchard et al. 2005). Thus, length-based management is a common approach to protect fish stocks from overfishing and to meet social and ecological objectives (Lewin et al. 2006). An increased consideration of size-specific harvest has also been proposed to help move single-species management towards the goals of an ecosystem-based fisheries management (Froese et al. 2008). As larger fish are typically removed first, fishing can lead to changes in the size and age distribution of the exploited fish populations (Harvey et al. 2006; Genner et al. 2010; Lewin et al. 2006, De Castro et al. 2015). Truncated size distributions can have detrimental effects as large and old fish are important for recruitment and population replenishment (Beldade et al. 2012; Barneche et al. 2018). Fishes also pass marked diet shifts as they grow. Larger individuals have a higher tendency to be piscivorous and thus may have a different function in the ecosystem. Removing large predators can have far-reaching consequences for the structure of ecosystems (Frank et al. 2005). Thus, preservation of large fish should be a key aim for fisheries management (Birkeland & Dayton 2005).

To this end, the frequently used management tool of minimum size limits can be effective (Ricker 1945; Jensen 1981; Maceina et al. 1998). The benefits of adopting also a maximum size, i.e. harvest slots, are increasingly shown to be even more favourable than minimum size limits, especially when mangers aim to balance potentially conflicting goals of conservation (e.g. natural age- and size structure) and fisheries objectives (high yield), e.g. (Jensen 1981; Gwinn et al. 2015; Tiainen et al. 2017; Ahrens et al. 2020). In a Swedish context, harvest slots can primarily be implemented for angling (handheld gears) and passive gears where the catch can be released alive. For commercial fisheries, the size composition of the catch can instead be influenced by minimum/maximum legal mesh sizes and associated effects on length selectivity, which can differ among species.

Length-based indicators (LBI) are widely used in aquatic systems to provide insights into fish populations' size structure (Fitzgerald et al. 2018; Kell et al. 2022). A representative sample of the length distribution of a fish population is enough to estimate LBIs, which can in turn reflect the impacts of size-selective fishing pressure (Sundblad et al. 2020). Additional, and more informative, LBIs can be estimated when additional data on length at maturity (L_{mat}; length at which 50% of females mature) and age to estimate growth parameters (e.g. L_{inf} or L-infinity; the size at which the average fish stop growing) are available (Froese 2004).

Reliable and simple indicators that are fast, easy and cheap to implement would provide useful tools for monitoring stock status and increase the likelihood that stocks can be continuously assessed. Simple indicators can also be collected and adopted by the fishers themselves, providing early warning signals and the need for more in-depth surveys and analyses. However, because length-based indicators respond not only to fisheries activities but also to environmental factors, a combination of several length-based indicators can provide better guidance for interpretation of stock status and effective management (e.g. Shin et al., 2005).

4.1. Formulation of objectives and targets

"Natural/Nature-like size structure"

There is no clear management goal regarding size distributions on a national level in Sweden. The responsible authority SwAM had such a fisheries management goal in their business strategy 2018-2020, which stated that managed stocks should have a size- and age-structure that maintain ecosystem functions ("Bestånden ska […] ha en storleks- och åldersstruktur som upprätthåller ekosystemens funktioner", sid 40 HaV verksamhetsstrategi 2018-2020). Although this formulation has been removed in the current strategy (2021-2023) , the general aim to maintain sustainable age- and size structures is still captured in management processes through the indicator C.3.3 of the MSFD and in the "measure" (åtgärd) 35 from the marine strategy for the North Sea and the Baltic Sea (HaV 2021, page 281). Yet there is no equivalent goal or indicator for freshwater fisheries, at least on the national level. Age/size structure, however, is a mandatory indicator when assessing ecological status according to the Water Framework Directive in lakes and is thus part of the EQR8 index. This index, however, do not apply to the large lakes included in this report. To the best of our knowledge there is currently only one regional example where management goals of natural size- and age-structures are explicitly stated – in Lake Vättern (Vätternvårdsförbundet 2017; Bryhn et al. 2021).

When formulating management goals and objectives for a size-based management, the long-standing work of (not the least) Rainer Froese can provide inspiration. A paper from 2004 provides a good example of three simple management goals related to size, including a framework with indicators and suggested target levels, to avoid overexploitation (Froese 2004; Cope & Punt 2009):

1. Let them spawn
2. Let them grow
3. Let the mega-spawners live

These goals can alternatively be formulated as goals that aim to preserve recruitment, optimize the fishery, and to preserve large fish and maintain a healthy size structure.

The first goal requires information on length at maturity (L_{mat}), i.e. at what size (or age) the majority of fish have reached sexual maturity, which reflects the reproductive ability. The general goal (for the fisheries) is that only fish above L_{mat} (as a reference point) are caught. Theoretically, an appropriate number of fish is supposed to be mature and spawn before being caught to maintain sustainable exploitation (e.g. Die and Caddy, 1997). To monitor this goal, data on maturity and catch composition is required. However, maturity information is lacking for many national stocks as they are not routinely assessed in monitoring and can also be difficult to determine. There is also growing evidence that standard histological gonad inspections are underestimating the size when fishes actually start to spawn and take part in spawning aggregations (Prince et al. 2022).

The second goal is related to an MSY-approach in the sense that fishing should use gears or otherwise be size selective so that the catch is mainly within the size interval of optimal length (L_{opt}), which is the length when the biomass of a cohort is maximized (Froese 2004; Froese et al. 2008). The target for the indicator when applied to a fisheries is that 100% of the specimens in the catch should be of optimum length. The size range employed has been proposed to be L_{opt} ±10%. In practice, the concept of L_{opt} thus shares the basic idea of harvest slots, where fishing is only allowed between a minimum and a maximum size. The use of harvest slots can also be considered to encapsulate all three goals above, as both recruits and large fish are excluded from the harvest. The use of harvest slots and fishing at L_{opt} has been proposed to outperform classical minimum-length limits and ensuring high catch and profit while maintaining large population sizes (Cardinale & Hjelm 2012; Gwinn et al. 2015; Froese et al. 2016; Ahrens et al. 2020). Although theoretically sound, a practical difficulty is determining suitable sizes and more quantitatively assessing stock status in these data-poor situations (Cope & Punt

2009). The L_{opt} approach has also received criticism for ignoring the importance of reducing fisheries mortality and for certain assumptions to be overly optimistic and not properly taking density dependent factors into account (Svedäng 2015).

The third goal is a more direct length-based goal or objective in the sense that it can be described by length-only indicators (details below). However, under the framework of Froese (2004), mega-spawners are defined as larger than L_{opt} +10%, which require information from growth models and natural mortality estimates. From a management perspective, it can be considered as a conservation goal, an ecological goal and a fisheries goal. It is a conservation goal in the sense of preserving large fish that are important for fecundity and spawning potential (Barneche et al. 2018); an ecological goal in the sense that the largest fishes have a unique role in the food-webs (Casini et al. 2009); and a fisheries goal in the sense that it can be used as an indicator of, or for, the fisheries – at least for catch and release fisheries targeting trophy-sized fish (Richardson et al. 2006; Bergström et al. 2022).

4.2. How do the indicators work?

Size- and age-based indicators can be divided into two groups, those that rely solely on length measurements and those that additionally include life-history parameters such as maturity and growth (i.e. parameters that rely on age data). An additional benefit of life-history parameters is also that they enable generalization possibilities as well as scaling of indicators, making them more comparable across stocks (Table 2).

4.2.1. Length-only indicators

A length distribution has unique information, as it describes the number, or proportion, of individuals of different sizes. When plotted over time it can effectively visualize changes (Figure 16). This type of information can then easily be communicated to stakeholders and managers.

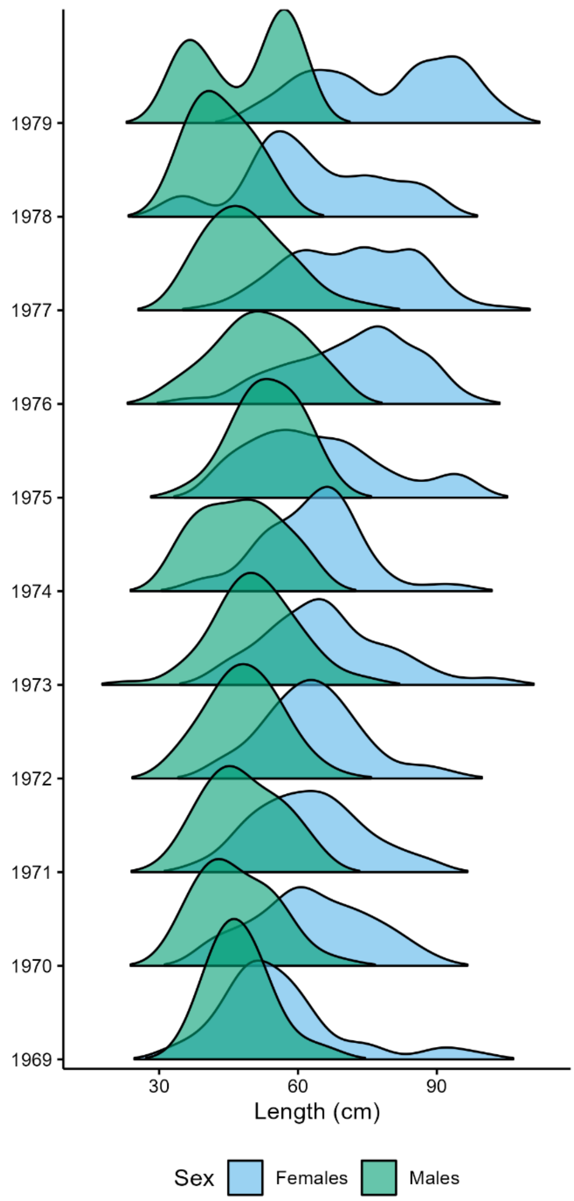


Figure 16. Example of a recovering length distribution over time (1969-1979) where the proportion of large females, but not males, increases over time. Data is from a former monitoring programme aimed at spawning aggregations of Northern pike in Lake Mälaren and the total number of pikes per year 1969-1979 was 115, 288, 129, 234, 159, 97, 67, 74, 204, 47, and 26 (unpublished, but see (Svärdson & Molin 1968)).

The simplest indicators can be derived from percentiles of a length distribution. Example indicators include the percentiles L10, L50, L90 or L95, where L10 is the length at which 10% of the fish in a sample are below this specific length. The quantiles describe different aspects of the size distribution. L10 can be considered to monitor the "recruitment potential" for future catches and should not be close to the median value (L50) as that could indicate recruitment failures. L90, or L95, describes the length of the largest individuals and should be well above the median

(L50). A decreasing L90 indicates increasing size-specific mortality or poor growth. Two things should be noted regarding the properties of the quantile indicators. Firstly, they are co-dependent. Since an entire size distribution is used as input for the calculation, changes in one end of the distribution impact the other indicators. Secondly, they are dependent on the selectivity of the gear and a prerequisite for their use is that the stock has been sampled in a representative manner. The practical use of L90 is perhaps best exemplified in the Baltic Sea, where it constitutes a core indicator for the size structure of coastal fish (HELCOM 2023b). To calculate L90 based on coastal monitoring data, a lower cut-off of 15 cm is first applied, which reduces the potential impact of recruitment fluctuations. Depending on the gear used, different threshold values (reference points) for good or poor status are applied. For Nordic multi-mesh gillnets and fyke nets, the threshold value to indicate good or poor indicator status is 25 cm, while 23 cm is applied on net series (Helcom 2023b; Bolund et al. in prep).

Another simple indicator is $L_{max5\%}$, which is the length of the largest 5% of the catch. An advantage of $L_{max5\%}$ is that it is not as affected by stochasticity compared to using only the largest individual (L_{max}). However, it can be calculated in several different ways. Note also that if $L_{max5\%}$ is applied to survey data it is more relevant for monitoring conservation goals, while if it is applied on fisheries (catch) data it can be used for both conservation as well as monitoring of fisheries goals/objectives (depending on management goals). An additional difference between applying it to survey versus catch data is the selectivity of the gears. Surveys seldom cover the largest individuals of the population. For example, multi-mesh gillnets have a lower probability of catching large fish than angling that targets trophy sized fish. The actual L_{max} values (and associated reference points) may therefore be very different depending on data source and should be applied accordingly in a management setting.

$L_{max5\%}$ was originally proposed by (Probst et al. 2013) and is calculated as "the mean total length of the observed largest 5% of the average number of individuals caught". The last part of that sentence is important. It means that based on a time series of length distributions, the indicator should be calculated on the average number of individuals included per year, thus including a fixed number of individuals in the calculations. This means that the indicator is different compared to using a yearly percentage, which leads to a varying number of replicates, as the quantile indicators do. This also means that the indicator primarily considers the right side of the length-frequency distribution, and is less impacted by recruitment variability (Probst et al. 2013).

Before moving on we can note that a recent paper showed how reference points for $L_{max5\%}$, consistent with a spawning potential ratio of 40% (Hordyk et al. 2015), can be obtained (Miethe et al. 2019). Although the main assumptions of that modelling framework are not likely to be perfectly fulfilled, i.e. constant recruitment, constant natural mortality and constant fishing mortality at length above L_c (where L_c is length at first catch), the study illustrates the ongoing efforts to perform analytical assessments on data-poor species (Kell et al. 2022).

Both L90 and $L_{max5\%}$ have been proposed to be scaled by L_{inf} (Fitzgerald et al. 2018). The benefits of such an approach are that it enables comparisons across species and stocks. A limit of $L_{max5\%}$ / L_{inf} >0.8 has also been proposed as a suitable reference point for good status (Fitzgerald et al. 2018; Kell et al. 2022). However, the calculation of L_{inf} requires additional information besides length composition.

4.2.2. LBIs with additional information

Additional life-history information that expands the use of length-only indicators include mortality, maturity and growth. Such information allows both for additional indicators to be calculated, as well as providing well-defined reference points to which the indicators can be compared (Fitzgerald et al. 2018; Kell et al. 2022) (Table 2).

Mortality is covered in another chapter of this report, and we here focus on maturity and in particular: growth-related parameters. To fit growth models, length and age data are necessary. Age is not routinely assessed in national monitoring and typical of data poor species we often have to rely on small and opportunistically collected data sets from either surveys or commercial sampling. Three growth models that are often applied to such data are:

- Von Bertalanffy; length = L_{inf} * (1 – exp(-(K * (age – t_0))))
- Gompertz; length = L_{inf} * exp(-exp(-K * (age – t_0)))
- Logistic; length = L_{inf} / (1 + exp(-K * (age – t_0)))

Where L_{inf} is the average maximum length of the oldest individuals (length at infinity, mm), which is formally defined as the asymptotic length where the growth rate is zero (for an average individual in the population), K is growth rate (mm/year) and t_0 is the theoretical age where size is zero. See e.g. (Nelson et al., 2009) for how to implement these in R.

Even though the growth parameters have been estimated from a small and separate data set, they can be more widely applied on length frequency distributions from other areas or years (under the assumption that these growth models are representative and stable for the assessed stock and time period).

Table 2. Summary of length-based indicators. For each indicator we highlight the primary data sou… it is applied and a short explanation. RP = reference point (sometimes referred to as threshold v… indicate the need to define stock specific RPs. Sources primarily used for the table are (Greenstree… 2012; Fitzgerald et al. 2018; ICES 2022; Kell et al. 2022).

Indicator	**Applied to**	**Explanation**	**RP**
L95, L90, L50, L10	Survey/catch data	Percentiles of the length distribution. Simple metrics to quantitatively describe a size frequency distribution.	ss
$L_{max5\%}$	Survey/catch data	Mean length of the largest 5%. Interpretation and reference points varies with source data.	ss
Size-slope	Survey data	The slope of the relationship between (log) abundance and length size classes. Steeper slopes indicate poorer conditions. Compare catch-curve analysis.	ss
L_{opt}	Catch data	Size when the biomass of a cohort is largest. $L_{opt} = L_{inf} * (3/(3 + M/K))$. If M is, or is assumed to be, 50% larger than K, the equation can be simplified as $2/3*L_{inf}$.	ss
L_{inf}	Survey data	Average maximum length of the oldest individuals. Most fish grow throughout life but growth decreases with age. L_{inf} is the asymptotic length where growth is zero.	ss
L_{mat}	Survey data	Length where (normally) 50% of the population reach maturity (by sex). Decreasing size at maturity can be a warning signal.	ss
L_c	Catch data	Length at first catch. Find the mode of the length distribution (length class with highest catch numbers Nmax), find first length class where catch is at or above Nmax/2. This is the length at first capture (Lc).	ss
mL_c	Catch data	Mean length of catch above Lc (i.e. observed mean when only including fish >Lc). The average size of the catch can function as a broad descriptor of the catch size composition and is also used in subsequent indicators.	ss
$L_{95\ or\ L90}/L_{inf}$	Survey/catch data	95th or 90th percentile of length distribution divided by Linf. The higher the value, the closer the observed value of L95 or L90 is to the theoretical average maximum length (L_{inf}). Reference point dependent on management goals and source data.	>0.8

$L_{max5\%}/L_{inf}$	Survey/catch data	Mean length of the largest 5% divided by Linf. The higher the value, the closer the largest fish is to theoretical average maximum length (L_{inf}). Reference point dependent on management goals and source data.	>0.8
P_{mega}	Survey/catch data	Proportion of individuals above Lmega = Lopt+10%. Generally catch of fish larger than Lmega should be avoided & Pmega in the population should be at least 30%. Reference point dependent on management goals.	>0.3
P_{mat}	Catch data	Proportion of mature individuals in catch. Target 100% in catch data.	≈1
$L_{25\%}/L_{mat}$	Catch data	25th percentile of length distribution divided by length at maturity. Assesses if the smallest individuals in the catch have reached maturity.	>0.3
L_c/L_{mat}	Catch data	Length at catch divided by length at maturity. Assesses if catch includes immature individuals.	>1
mL/L_{mat}	Catch data	Mean length divided by length at maturity. Assesses mean length of the catch in relation to size at maturity.	>1
mL_c/L_{opt}	Catch data	Mean length of individuals >Lc divided by optimum size for balancing biomass production of stock with natural mortality. Aim is to ensure that fishing is primarily directed at optimal sizes.	≈1
P_{opt}	Catch data	Proportion of fish caught within +-10% of Lopt.	≈1
mL_c/LF	Catch data	Mean length of individuals larger than Lc dived by LF, where LF = M = (0.75Lc + 0.25Linf). This indicator is the "f" component in the "rfb rule" applied on category 3 species in Ices 2022. Note, the reference length (LF) assumes M/k = 1.5.	≈1
Large Fish Index	Survey data	Percent biomass of all species >Xcm in the sample. Community focused indicator.	ss
Mean of max	Survey data	Mean of maximum length of all species in the sample. Community focused indicator.	ss

An analytical framework that uses similar life-history parameters as used for some of the indicators above is the ‘length based spawning potential ratio’ (LB-SPR), which was recently tested on Baltic Sea pike (Fitzgerald et al. 2023). The general concept is based on assessing the spawning potential of a fished stock in relation to the expected composition in an unfished state, based on length composition in the catch and life-history parameters (Prince 2003; Prince et al. 2011, 2020; Hordyk et al. 2015, 2016). The method thus yields an assessment of the spawning potential expressed as a percentage, where lower values indicate poorer state. A target level generally adopted for management is a SPR of 30-40%, and a proposed limit reference point to stay above 20% (Prince et al. 2015). Besides, a catch length composition on which to apply the model, key parameters are i) the M/K ratio (natural mortality over the growth parameter k), ii) mean asymptotic length (L_{inf}), iii) the variability of length-at-age (CV_{Linf}, often assumed to be 10%), and iv) size at maturity (L_{mat}, for both 50% and 95% mature).

The ratio L_{mat}/L_{inf} from other studies can be used to estimate L_{inf} in situations where it is unknown (Prince et al. 2020). The LB-SPR method relies on relationships between the size structure of the stock and the relative fishing pressure (F/M) and the two ratios (M/K and L_{mat}/L_{inf}). Based on maximum likelihood, the models simultaneously estimate the size at which individuals in a stock become vulnerable to capture and the relative fishing mortality (F/M), which are then used to calculate the SPR. Capture vulnerability is related to size at catch (L_c). The model assumes a logistic selectivity curve using the selectivity-at-length parameters SL50 and SL95, i.e. the sizes with 50% and 95% probability of being caught by the gear. The underlying assumptions of the model are asymptotic selectivity, adequately described growth parameters, length-at-age is normally distributed, natural mortality rates are constant across adult age classes and that growth rates remain constant across the cohorts within a stock (see references above). The resulting estimates of SPR is highly influenced by the size of the largest individuals in the sample relative to L_{inf}, as well as to L_{mat}. Thus, underestimation of L_{inf}, as well as large and fast changes in recruitment can have a strong impact, as shown in simulation studies (Hordyk et al. 2015). An extension of LB-SPR adding dome-shaped selectivity patterns has also been developed by (Hommik et al. 2020).

4.3. Application to the selected stocks

To illustrate a subset of the indicators we apply them on Northern pike in Lake Vänern.

4.3.1. Northern pike in Lake Vänern

The following section will illustrate different growth models and the calculation of various indicators applied to pike in Lake Vänern. Typical of data poor species there is only a small and opportunistically collected data set. In this case, growth models are fitted to a small sample from commercial fisheries in 2014 (Figure 17). We then applied the growth models and selected indicators on another dataset containing length information from recreational fisheries, collected annually by the Swedish Anglers Association 2014-2021. As pike have sex-specific growth and the fisheries primarily target large fish, we use female-only growth models, N=39 (Figure 17).

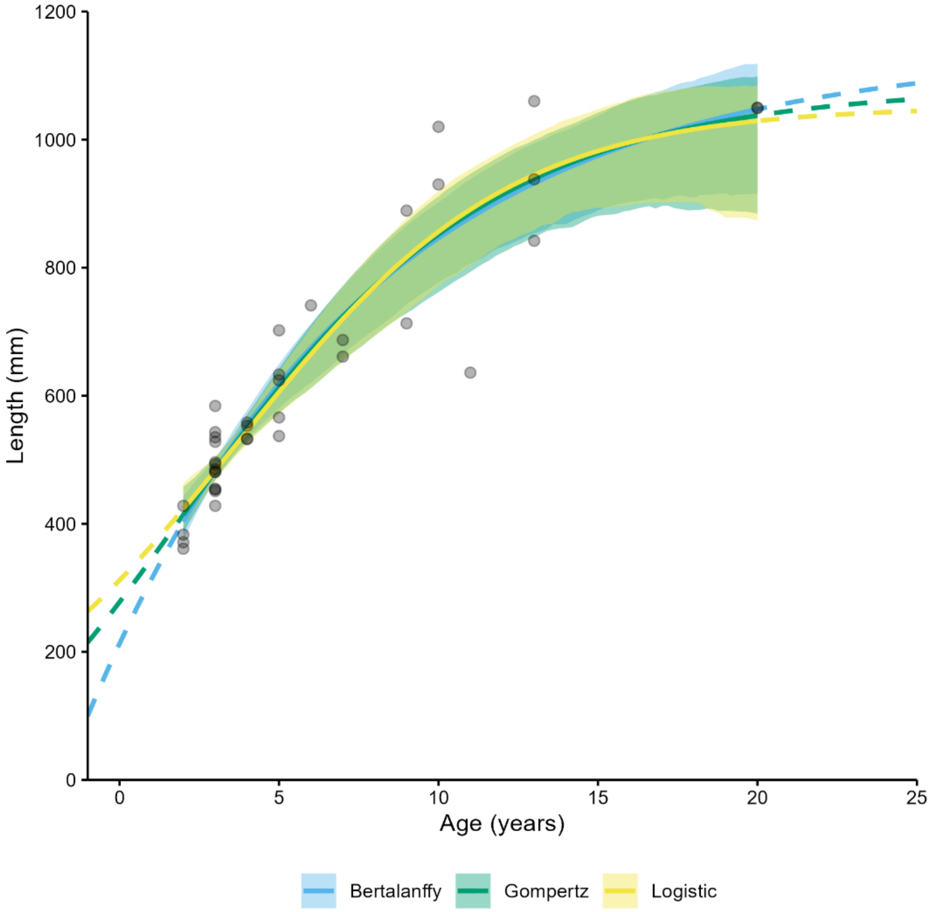


Figure 17. Three growth models (lines) and associated uncertainty (bands) applied to female Northern pike (N=39, grey dots), collected from commercial fishers in 2014. Note that there are two individuals aged 20 years. Models are extrapolated outside the data range (dotted lines) to highlight differences between models.

Although the sample size was very small, all three growth models had an acceptable fit. Especially the two oldest individuals (both aged 20) were important for the fit of the model. Parameter estimates were similar among the models, with overlapping confidence bands (Figure 18). Under the assumption that these growth models are representative and stable, we can use the parameter estimates to calculate LBIs as long as we have information on length (Figure 19). The assumption that the growth models are representative can be critical. For example, if growth is calculated from populations already affected by harvest, leading to a loss of large and old individuals, there is a risk that L_{inf} is underestimated. However, L_{inf} around 100 cm is a reasonable size for healthy pike stocks.

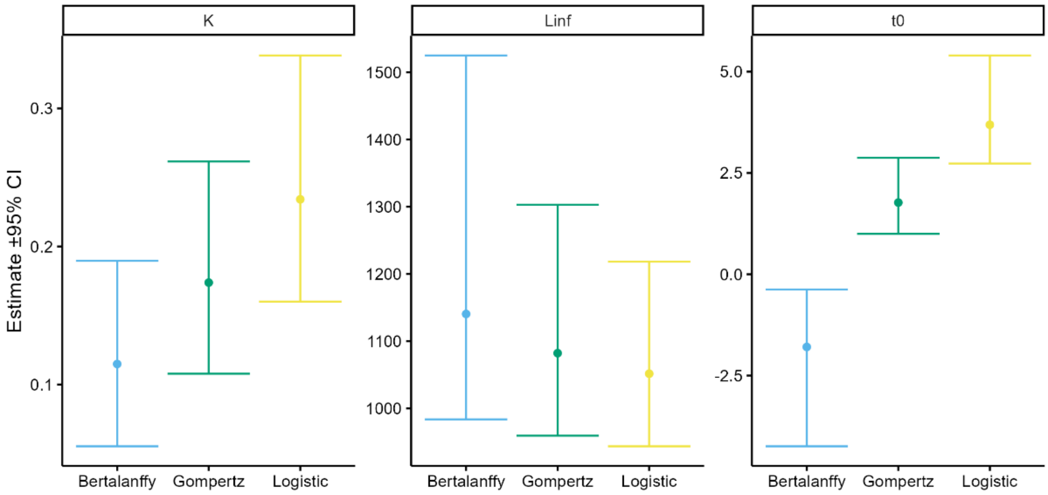


Figure 18. Parameter estimates and associated uncertainty for the three growth models in Figure 2.

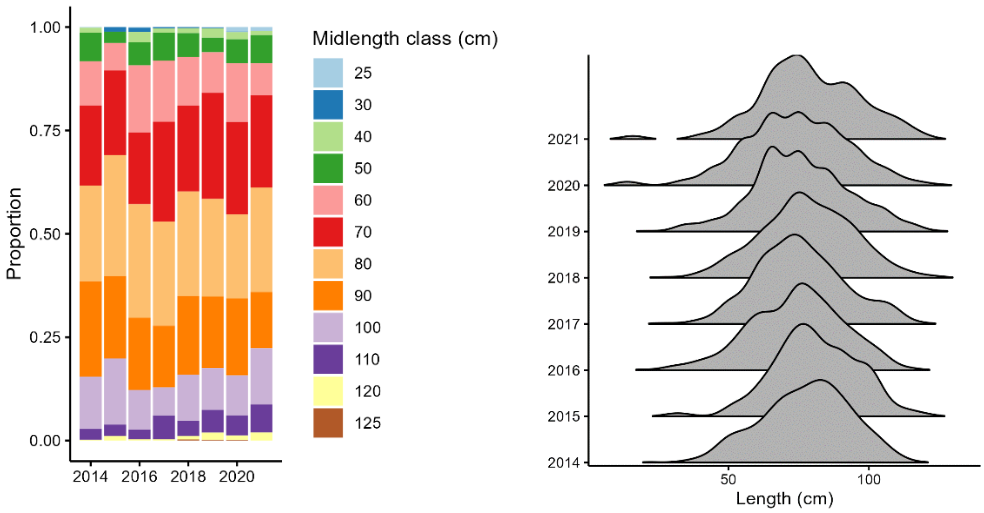


Figure 19. Proportion of pike per length class (left) and relative size frequency (right) per year. Both figures contain the same angler data and illustrate two different ways to communicate raw size frequency data from Lake Vänern.

Length-only indicators applied to Northern pike in Lake Vänern

Starting with the simplest quantile indicators that rely on length-only data, no clear trend is apparent (Figure 20). Applying $L90/L_{inf}$, yields values >0.8 for all years and growth models (data not shown).

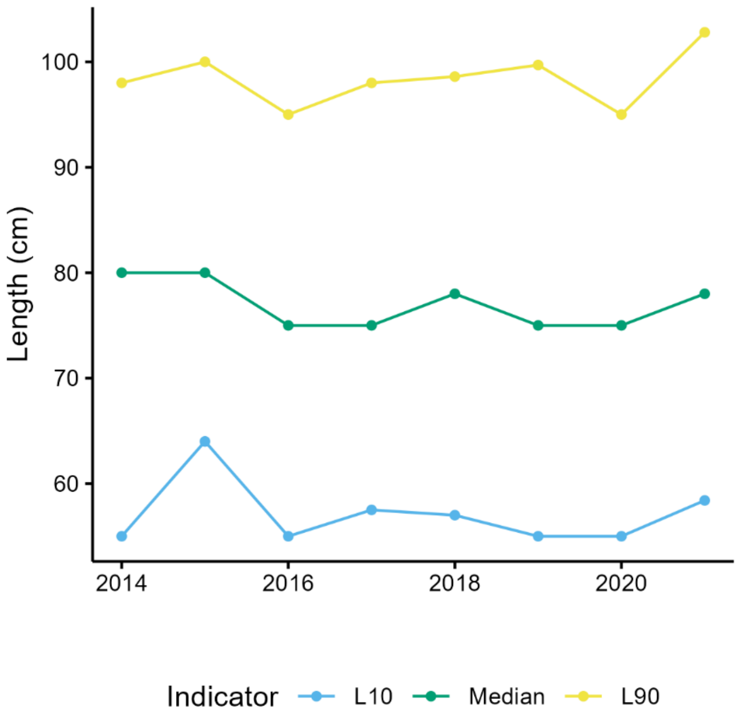


Figure 20. Quantile length-only indicators applied on the Northern pike angler dataset from Lake Vänern.

Another length-only indicator is the $L_{max5\%}$, which is the average length of the 5% largest individuals in the sample. Again, let's examine the indicator properties using the L. Vänern angler app data. First, we calculate the number of reported pikes in the app per year (Table 3).

Table 3. Total number of reported pikes in the angler app data from Lake Vänern per year, and number of pikes if 10% or 5% are selected per year.

Year	Total number	N 10%	N 5%
2014	569	57	28
2015	181	18	9
2016	572	57	29
2017	576	58	29
2018	815	82	41
2019	1824	182	91
2020	640	64	32
2021	103	10	5

Notice the range, 103 – 1 824 individuals, and the large variability between years. As will be shown, such a large variability can have a large influence on the indicators. The average across years is 660 and 5% of that is 33 individuals. We therefore calculate the average length of the largest 33 individuals per year. Notice that it is quite a high percentage of sampled fish in 2021 and 2015. We call this indicator L_{max33N}, as it is calculated on 33 individuals per year (Figure 21). To get an estimate of uncertainty around the indicator a bootstrap analysis is performed.

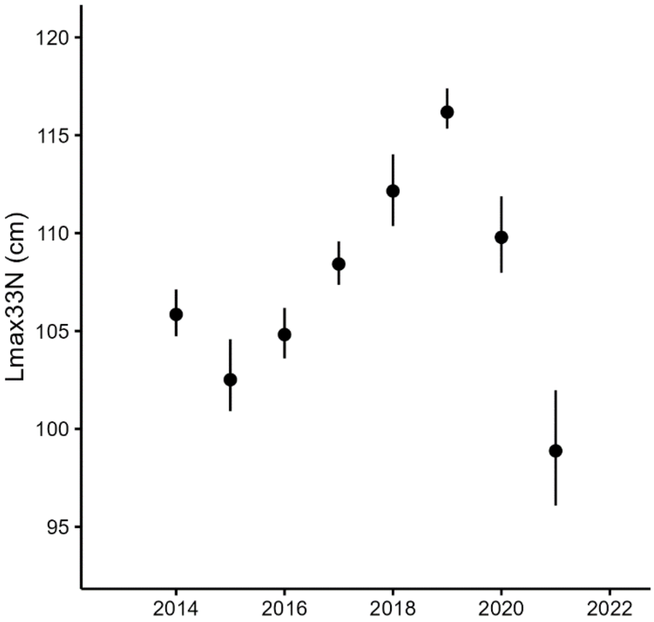


Figure 21. Lmax5% calculated according to (Probst et al. 2013), i.e. with the fixed number of 33 individuals per year, which is 5% of the average number of reported pikes across years. Uncertainty is 95% confidence interval estimated by bootstrap.

When compared to the raw data plots above (Figure 19), the result is not as expected (Figure 21). The year 2021 indicates a powerful reduction, in direct conflict with the apparent increase of large fish when plotting raw data. This is a consequence of the variability in the number of reported pikes per year.

Let's compare L_{max33N} with two alternative calculations of Lmax using a smaller number of fish. This should reduce the risk of including too large parts of the left side of the size frequency distribution when the sample size is small. We call these alternative indicators L_{max5} and L_{max10} and try both as a percentage per year (PC) and as a fixed number per year (N). The number of individuals used for the calculation per year is 10-182 for 10% and half (5-91) for 5% (Table 3).

A fixed number (Method = N) tends to yield higher L_{max} values (Figure 22). This is not surprising since the percentage method (Method = PC) in most cases have a higher number of fish included, thus including more of the left side of the size frequency distribution. This is nicely illustrated by the year 2021, when 103 fish were reported, thus making the PC and N method identical. A second lesson learnt is that the N-method tends to have higher variability (years 2018-2020) while the PC-method can be considered less variable, and more in line with the raw data plots shown earlier. Thirdly, the L_{max} indicator is here presented both as L_{max5} and L_{max10} in both percentage and number of fish. It is apparent that trends are very similar, but that absolute values differ somewhat. This suggests that the choice of L_{max5} or L_{max10} regardless of if they are expressed as a percentage or fixed number, is of lesser concern for trend analyses (as long as "enough" individuals have been sampled). Lastly, not only the number of samples per year but also the large variability in the number of samples per year plays an important role for these indicators. We recommend that an analyst using L_{max} (as well as L10, L50 and L90) carefully consider adequate sample sizes from which to calculate the indicators. Regarding L_{max}, we generally recommend that $L_{max}5$ as a percentage per year is used.

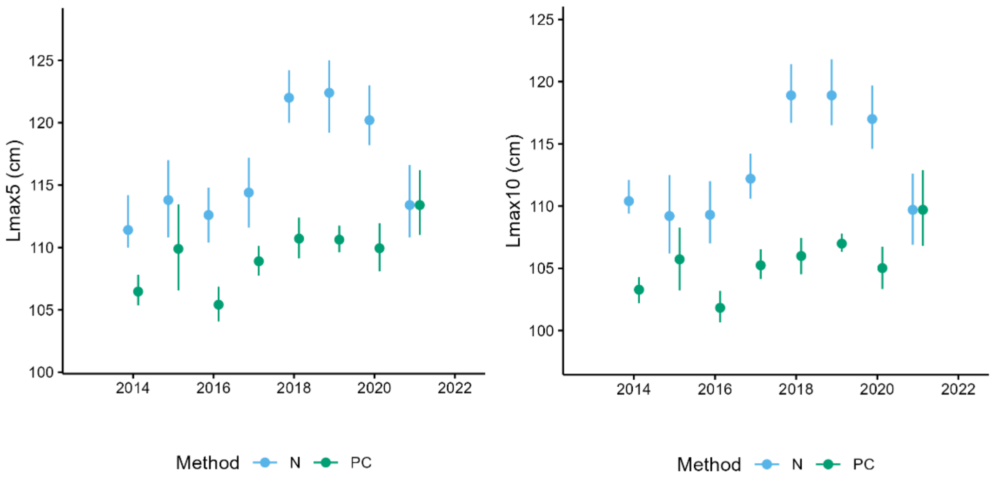


Figure 22. The length-only indicator L_{max5} (left) compared to L_{max10} (right) using either a fixed number of individuals (method=n) or a percentage of individuals (method=pc).

L_{opt}, M and related indicators with additional life-history information

The size at which the biomass of a cohort is at its maximum (L_{opt}) is calculated as $L_{opt} = L_{inf} * (3 / (3 + M/K)$, from (Beverton 1992; Froese 2004). However, as noted in the mortality chapter, natural mortality (M) can be difficult to calculate and it can be considered prudent to check how L_{opt} depends on the uncertainty of the M-estimation.

For Northern pike, the relationship between L_{opt} and M was examined by using the three estimates of growth across a range of M-values (Figure 23). Within each growth model, changes in M can lead to relatively large changes in L_{opt}, especially under von Bertalanffy-growth (Figure 23).

Point estimates of M was calculated according to equation 14 in (Miethe et al. 2019), originally from (Then et al. 2015, 2018):

$$M = 8.87K^{0.73}L_{inf}^{-0.33}$$

Where M is the natural mortality, K is the growth parameter and L_{inf} is the asymptotic size. Note that the equation refers to L_{inf} in mm (if it is in cm, the 2015 paper applies).

Applying this equation to Lake Vänern pike yielded M=0.18 for von Bertalanffy (Linf =1141 mm, K=0.11), M=0.25 for Gompertz (Linf = 1082, K=0.17) and M=0.31 for Logistic growth (Linf = 1052, K=0.23). Despite differences in the M-estimates, the three growth models yield similar L_{opt} -values, which is good yet not surprising as L_{opt} is scaled to both K and L_{inf} (Figure 23).

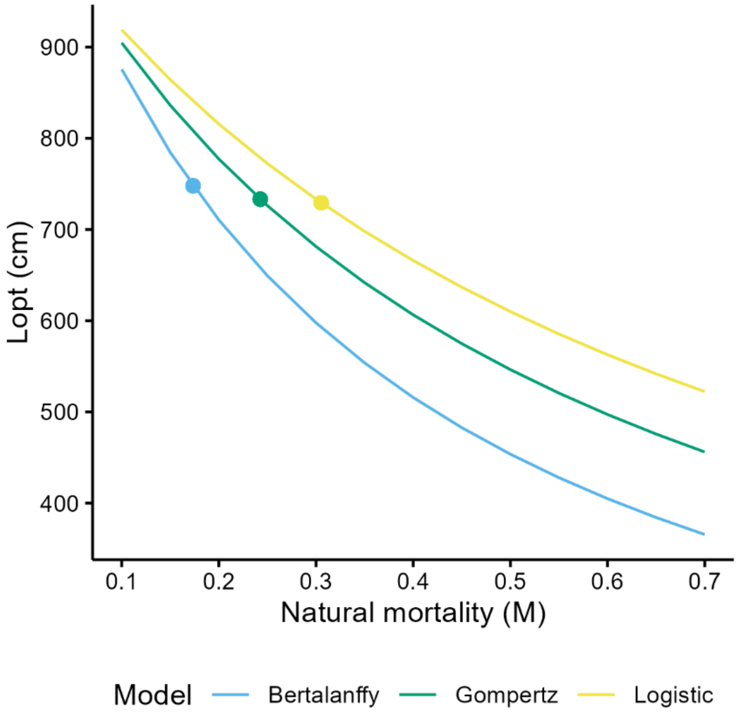


Figure 23. L_{opt} as a function of natural mortality (M) using the parameter estimates from the three growth models for Northern pike in Lake Vänern. Points are the M-values calculated from Eq 14 in (Miethe et al. 2019).

The average L_{opt} (value=734 mm, based on point estimates of M) can then be used to calculate additional length-based indicators, such as P_{mega} and P_{opt} (Table 2). P_{mega} is the proportion of mega-spawners, which should be high in survey data. Here, we apply it to catch data from a catch-and-release trophy fishery, which can then be thought to fulfil not only conservation goals (as in survey data) but also fisheries goals (targeting trophy sized fish). The proposed reference point for P_{mega} is 0.3 (Froese 2004; Fitzgerald et al. 2018) and it appears well suited to Lake Vänern pike (Figure 24).

P_{opt} is the proportion of fish at L_{opt}±10% and relates to management goals of optimal yield (MSY-related resource use) by ensuring that the fisheries target optimal lengths. Therefore, P_{opt} is only relevant for catch data, not survey data. The goal is that 100% of the catch is at optimal length, thus the goal is that P_{opt} is about equal to 1, at least in a catch-and-kill fishery. When applied to this dataset, P_{opt} is around 0.3, indicating that only a third of the catch is at optimal length, far from the goal of 1. This reflects the fisheries size selectivity and how the largest individuals are particularly targeted (and caught). However, as the data comes from sport fishing (angling) that to a very large extent practice catch and release, it is of less concern for the status assessment. Had it been a catch-and-kill fishery, management should have needed to adapt regulations to change the size structure of the catch (e.g. changing mesh sizes or length limits).

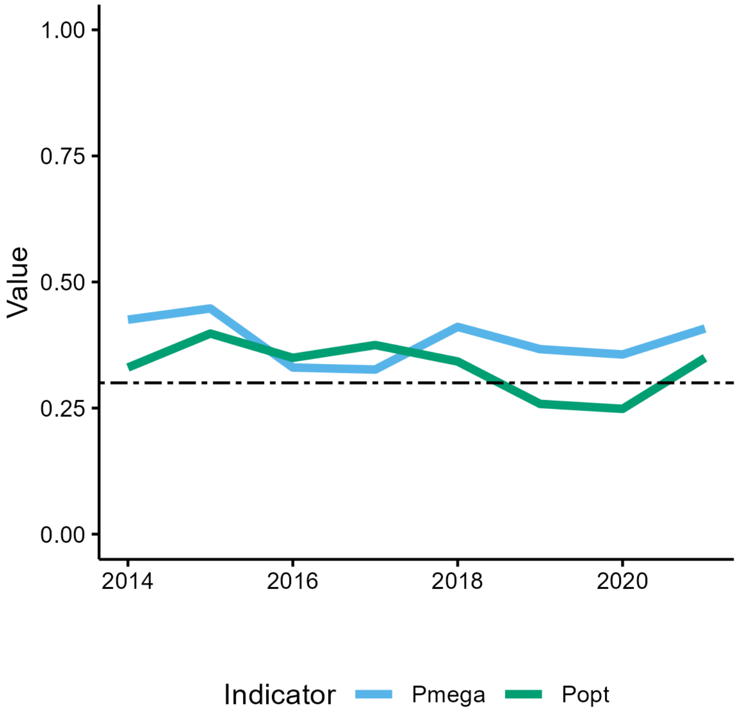


Figure 24. Indicators P_{mega} and P_{opt} applied to the angler app data on Northern pike in Lake Vänern. Dashed line indicates the proposed reference level (Froese 2004; Fitzgerald et al. 2018).

4.4. Challenges and lessons learned

As national stocks in many cases can be classified as data poor, we often have to find solutions by adopting the “something old, something new, something borrowed” song. As illustrated above, the growth data for Northern pike was “borrowed” from commercial sampling in a specific site on a specific year and applied to the angling catches length distribution from across the entire lake.

Although this was within the same lake and approximately the same time period, it can be appealing to borrow life-history variables from other stocks, or even related species. However, such approaches should be used with care as many parameters can vary, as shown for example for growth in bream (Sundblad et al. 2020). Nevertheless, for large predatory fish that are targeted by angling (recreational fisheries), local ecological knowledge on expected, or preferred, maximum sizes (L_{inf}) can often be derived. Such information could potentially be applied elsewhere, e.g. on stocks that have truncated size distributions and lacking large, fast-growing individuals – for example exporting L_{inf} from Lake Vänern (or other freshwaters with healthy pike stocks) to other areas where the stocks are in poor condition (for example Swedish Baltic coastal areas, Bergström et al., 2022; Olsson et al., 2023). However, as growth rates, maturity and other life-history parameters can be dynamic, applying "old" or "borrowed" estimates may be erroneous and lead to incorrect status assessments.

A limitation with LBI's that rely on growth information is that they are to a large extent restricted to fish. For crustaceans, indicators that rely also on growth are difficult to use since crustaceans cannot be aged the same way as fishes. The lack of otoliths, or other permanent calcified structures used for age-determination, in combination with a step-wise growth related to molting means that growth is very different from fish. Although there are methods for estimating crustacean growth, for example by calculating the molting increments and inter-molt periods, such data require e.g. well designed mark-recapture experiments (Chang et al. 2012).

Another limitation, or at least aspect to consider when applying LBIs, is gear selectivity. Gears have different efficiency and potential of catching different sizes, and depending on their selectivity, an accurate representation of the size distribution may or may not be obtainable (HELCOM 2023b). Nevertheless, it is not always necessary to have a perfect representation of the size distribution, especially if the indicators are assessed with data from the same gear used over time, as temporal changes may still be monitored. However, ensuring representative gears and comparable selectivity is important for all length-based approaches.

As noted above (Table 2), most indicators have been developed for catch data, i.e. sampled from the fishery. As illustrated with pike (above) and perch (HELCOM 2023b), they can often be applied also on survey data. However, the interpretation and reference points may then be very different. For example, survey data can be used to monitor conservation of immature individuals, while the proportion of immature individuals should be low in catch data. Similarly, if the aim is to conserve large individuals and there is a harvest slot size in place, indicators of large fish should be low in catch data (as large fish are larger than the harvest slot and should not be caught), and high in survey data (to ensure that large fish are still existing). However, the proportion of large fish could potentially also be high in catch data, especially if the fishery is primarily a catch-and-release fishery with low release mortality since the large fish are then not harvested but released (as exemplified by pike above).

A potential problem with LBIs, especially when length data is collected opportunistically or outside standard monitoring protocols by trained personnel, is data quality. By quality, we here mean potential reporting bias, precision of length measurements and longevity (maintaining time series). Reporting bias could occur for example if the rapporteurs tend to report large fish more often than smaller sized fish, which could bias the length distribution towards larger sizes and falsely lead to inflated indicator values. Also, if length is reported in e.g. 5cm-bins, larger changes in the size distribution are required before they can be detected compared to if length is precisely measured in mm.

Yet another difficulty lies in obtaining reliable length-(or age)-at-maturity (L_{mat}) values, which are required for several indicators focusing on the left side of the length-frequency distribution. This problem was particularly pronounced for Northern pike in L. Vänern. Pike are mainly caught in commercial fisheries in spring in association with spawning migration. Thus, immature pike are nearly absent in the catch and any measure of L_{mat} is potentially biased.

4.5. Recommendations

For assessment purposes, we recommend that only methods and indicators that have been tested and validated are used (HELCOM 2023b). Similar to the other groups of indicators (mortality and abundance/biomass), length-based approaches (including those with additional information) may vary over time and between different areas, and thorough tests of their applicability increase the chances of better assessments.

Indicators that focus on the left-hand side of the length frequency distribution (e.g. L_c, L10, and L25%) can be impacted by strong recruitment events and therefore have high variability, which need to be considered when such indicators are used. The recommendation is therefore to primarily use length-based indicators on long-lived species and to use indicators that focus on the right side of the length-distribution and goals to conserve large fish, e.g. mL, L_{max}5%, L90, L95 and P_{mega} (Kell et al. 2022) and to evaluate truncation of the length structure in the stock (Shin et al. 2005; Rochet et al. 2010).

Since length-based indicators are generally better at estimating trends than states, i.e. absolute levels (Kell et al. 2022), it is recommended to ensure longevity in length data collection in order to obtain long time series. However, although requiring much work, it is possible to develop management reference points to assess absolute levels as well, as exemplified by L90 for coastal perch (HELCOM 2023b, Bolund et al. in prep). As reference levels have been developed for coastal perch, it may be tempting to "borrow" those also to other areas (e.g. the Swedish large lakes). But before doing so, thorough analyses are needed, as reference levels are generally stock specific (Kell et al. 2022).

Length based approaches that rely also on estimates of mortality, growth and maturity increase the number of potential indicators that can be calculated and also enable well-defined reference points (Table 2). These more "complicated" assessment methods, e.g. LB-SPR, can provide standardised frameworks and are thus appealing to apply also for nationally managed stocks in Sweden. However, as they rely on fisheries catch length compositions, and such data is very seldom collected from the fisheries, their implementation is hindered without dedicated data collection programs. Nevertheless, growth parameters and maturity can also function as indicators in themselves, and the collection of such data should be prioritised.

In summary, we propose that future work continue to develop methods in a case-specific order, i.e. for a particular stock; work out management goals, associate the most robust LBIs, identify appropriate references points, and ensure necessary data is collected (survey and/or catch data, including also maturity and age). The work should be coordinated with the development of regional management plans and co-management initiatives in participatory processes. Such processes, that create common goals and frameworks, can also be expected to aid the collaboration within the "interaction triangle" of managers, scientists and fishers (both commercial and recreational) (Röckmann et al. 2015). By utilizing modern techniques, simple reporting systems can provide necessary length-data to aid the assessment of data-poor species.

5. Development of a stock assessment model for pikeperch in L. Hjälmaren

5.1. Rationale

Analytical stock assessment models are routinely used within fisheries management. The ability of a stock assessment model to estimate spawning stock biomass, recruitment, harvest levels and reference points is desirable for any stock that is harvested commercially. Our ability to apply assessment models in national waters is often limited by data availability. However, some of the Swedish national stocks potentially have enough data for analytical assessment models to be applied. The data requirements for an analytical model vary with the model but generally include catch age composition, maturity, recruitment and the relationship between spawning stock biomass and recruitment, natural mortality, fisheries mortality, growth parameters, catch and effort in the fisheries, fisheries independent stock index, selectivity and length weight-relationships. Especially knowledge on the fisheries catch composition (length, age and selectivity) as well as time series of such data limit the general application of analytical models for national stocks. Here we report on the development of applying an assessment model to the pikeperch fishery in Lake Hjälmaren, a stock for which much of the data needed has been collected through dedicated programs.

5.2. Assessment model

The model chosen for assessment of pikeperch in Lake Hjälmaren is the Stock Synthesis (SS) model (Methot & Wetzel 2013). Stock Synthesis is an age- and size–structured stock assessment model that simulates fish growth, maturation, mortality and other biological processes (population sub-model), estimates values of parameters describing those and other processes (for example fisheries selectivity) from data sources (observation sub-model), estimates goodness of fit to the data in order to get best fitting parameters (statistical sub-model), and potentially projects quantities fulfilling management objectives (forecast sub-model).

5.3. Input data

An overview of the datasets included in the model is shown in Figure 25. Fisheries landings of pikeperch come from two types of gears, gillnets and trap nets, and thus are represented as two fleets. However, prior to 1996 total landings were reported without accounting for gear type, thus mean ratio between landings by gear in 1996-2021 was applied to separate total landings by gear before 1996. A minimum landing size regulation was implemented for the stock, which was 40 cm prior to 2001 and 45 cm from 2001.

Biological information (age-length-key (ALKs) and length distribution) associated with fisheries data was collected from only one commercial fisher and was assumed to be representative for all other fishers, since the fisheries is conducted in the central part of the lake and pikeperch in Lake Hjälmaren consists of one genetically homogeneous population (Dannewitz et al. 2010). Fish below landing size were removed from length distribution data from trap nets since they are discarded and assumed to survive (Nyberg et al. 1996). Length distribution from the gillnet fleet is available only from a single year.

The biomass index (CPUE, weight per net and night) used in the model originates from two sources. The recent gillnet survey (denoted gillnet_survey) is from monitoring data using the standard multi-mesh gillnet “Bkust9+2” at one site 2008-2011 and another site 2013, 2016 and 2019. Both periods were fished in the main basin Storhjälmaren but the site fished during 2008-2011 was generally shallower than the recent survey site and was characterized by a smaller mean fish size (not shown). The historical survey from the 1960’s (denoted gillnet_survey_h) is from the gear “Blänk” which consists of a series of nets with different mesh sizes linked together. The historical survey was performed at a third site in the shallow basin Mellanfjärden situated northwest from the main central basin. The stock assessment model assumes that the surveys are all representative of the same population. The differences among the survey sites and gears have initially been examined in a modelling framework, where we attempted to develop a joint stock index. In the model described in this chapter however, we use “raw” index data without accounting for these differences.

Individual fish data are from the same sources but not collected annually (Figure 25). Note that the recent length composition from the gillnet survey is represented as a single timepoint.

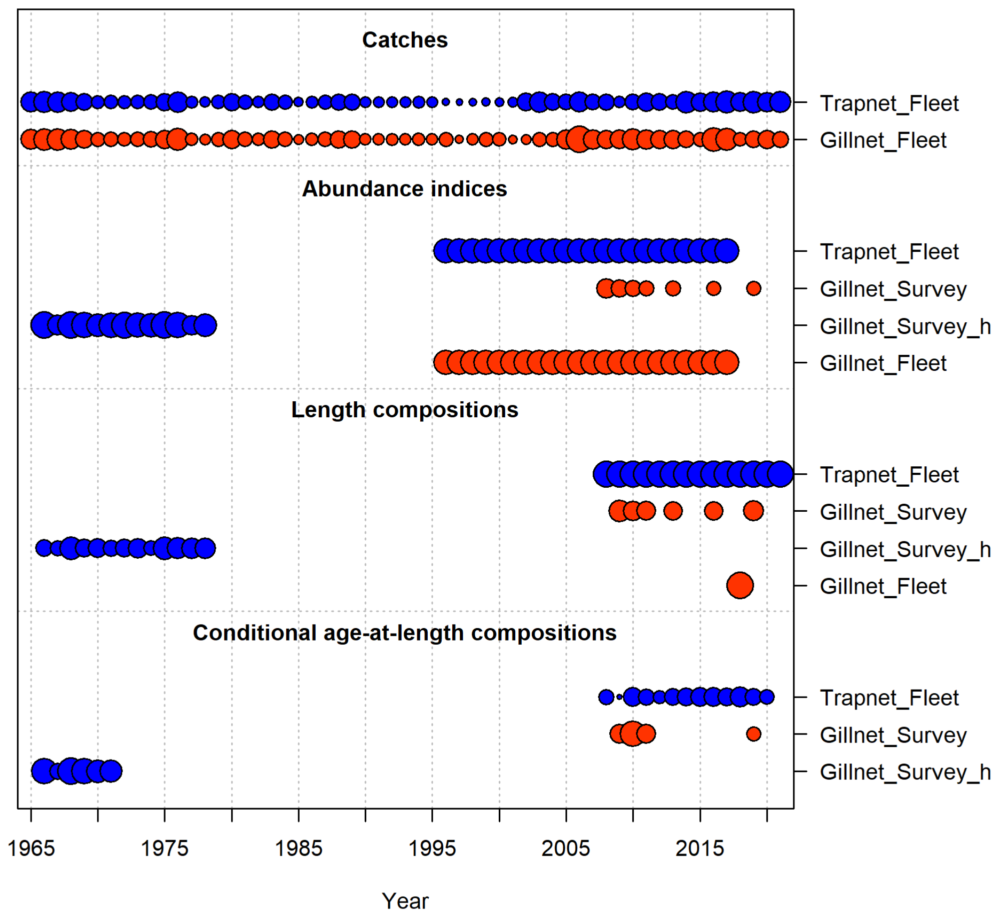


Figure 25. Summary of the input time series included in the model. Circles are proportional to total catch for catches; to precision for indices, discards, and mean body weight observations; and to total sample size for compositions and mean weight- or length-at-age observations. Note that since the circles are scaled relative to maximum within each type, the scaling within separate series should not be compared. See text for details.

5.3.1. Samples sizes, CVs, data weighting

The contribution of each data to the overall log-likelihood function in stock synthesis is inherently weighted by the measurement variance of that data, which is set by adjusting the CV. The CV thus represents how much flexibility the model has to deviate from the data, which can be used for weighting purposes. For the commercial fleet the coefficient of variation (CV) of the catches was set to 0.05. The CV of the initial catches of the commercial fleet was set to 0.1 to account for additional uncertainty. The annual sample size in the age-length-key (ALK) is the number of fish sampled.

The CV of the survey biomass indices in the historical and recent surveys was estimated as standard error (SE) and log-transformed as recommended in the Stock Synthesis manual (Methot & Wetzel 2013).

5.3.2. Assessment model formulation

The assessment model for pikeperch is a one area, annual, age-based model where the population is comprised of 14+ age-classes (with age 14 representing a plus group) with sexes combined (male and females are modelled together).

The model starts in 1966 and the initial population age structure was assumed to be in an exploited state, so that the initial catch was assumed to be the average catch in 1966-1968. Fishing mortality was modelled using the hybrid F method (Methot & Wetzel 2013). Option 5 was selected for the F report basis; this option represents fishing mortality requested by the ICES framework (i.e. a simple unweighted average of the F of the age classes chosen to represent F in the plots below, called Fbar (ages 4-6)). Fbar represents the average fisheries mortality for the age-classes that are fully exploited by the fisheries.

Spawning stock biomass and recruitment

Spawning biomass was estimated at the beginning of the year. In the model, the recruitment was assumed to be only a single event occurring at the beginning of the year. Recruitment was derived from a Beverton and Holt (BH) stock recruitment relationship (SRR) and variation in recruitment was estimated as deviations from the SRR. Recruitment deviates were estimated for 2008 to 2019 (12 annual deviations), since there was a gap in length distribution data in 1979-2007 and ALK in 1972-2007. Recruitment deviates were assumed to have a standard deviation (σ_R) of 0.5 and steepness (h) of 0.93 (estimated from life history parameters from Fishlife database)(Thorson 2020).

Growth, weight and maturity

Several parameters for the analytical model were estimated prior to fitting the stock synthesis model (Table 4, Appendix). Parameters included growth, which was modelled according to the von Bertalanffy equation (von Bertalanffy 1938), length-weight relationships, and maturity ogive. Input data was based on biological samples collected during surveys and from fisheries in 2008-2021. The growth parameter (k) in the growth function was re-estimated in the SS3-model during one of the preliminary runs.

Table 4. Growth, length-weight and maturity parameters used in the model. Linf is the asymptotic length, k is the growth parameter, Lmin is length at minimum age (0.5 years), alpha and beta are coefficients for the weight-length relationship, L50 is the length (cm) where 50% of the populations is assumed to be mature and slope is the estimate for the maturity relationship.

Parameter	Value	Estimation
Linf (cm)	70	outside model
k	0.15	estimated by the SS3 model
Lmin (cm)	13.9	outside model
alpha (weight-length)	2.06E-06	outside model
beta (weight-length)	3.36	outside model
L50 (maturity)	42.69	outside model
slope (maturity)	-0.175	outside model

Natural mortality

An age-varying natural mortality (M) is included in the model, which is assumed to be constant for the entire time series (Figure 26, Table 5). M was estimated based on the Chen and Watanabe method (Chen & Watanabe 1989), using the website “barefootecologist.com.au/shiny_m.html”. Estimation of M was based on the parameters of the von Bertalanffy growth function and assumes a decline of M with fish age. To reduce the number of parameters in the model, natural mortality was set using 4 break points: age 0.5, 1.5, 5 and 15, where M for the adjacent ages is linearly interpolated from estimated values at break points.

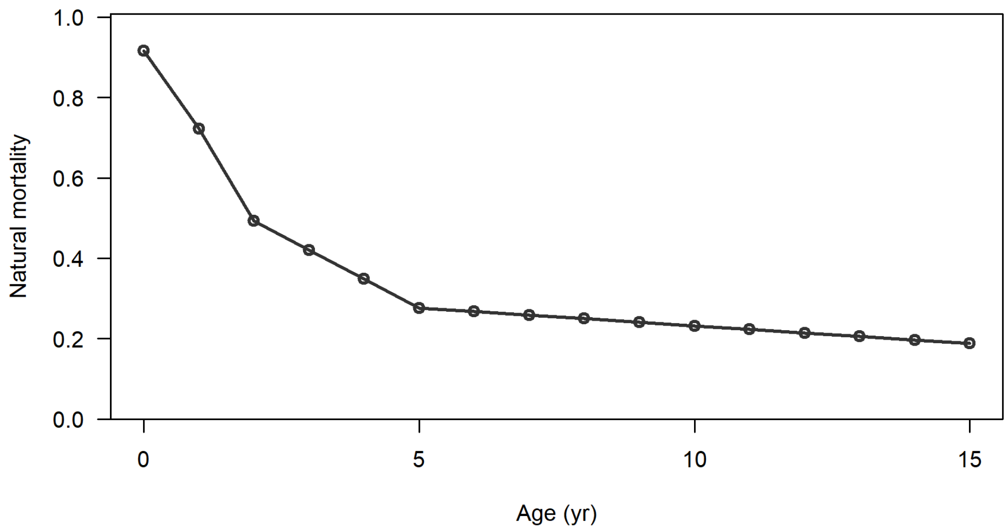


Figure 26. Age-specific natural mortality used in the stock assessment model.

Model settings

Fisheries and survey selectivity in the model is a function of length. Change in pikeperch fisheries regulation from a minimum landing size of 40 cm to 45 cm in 2001 was accounted for by introducing two time-blocks: 1966-2000 and 2001-2021; and fixed selectivity parameter defining the peak of the selectivity curve of the trapnet fleet to 40 and 45 cm correspondingly. Other parameters of the selectivity curve were also fixed in the model. In order to account for changes in the mesh size in gillnets, selectivity parameters (peak and ascend) were first estimated for the total time period (1966-2021) fitted to the length distribution from 2018. These selectivity parameters were then assigned to the first time-block 1966-2000 and estimated again for the period 2001-2021. As the selectivity parameters 'determine' the size distribution of the catch, and we assume that undersized fish caught in the gillnet are discarded and have a high mortality (100%), we haven't accounted for the change in minimum landing size for the gillnet fleet, thus allowing the gillnet fleet to include catch of undersized fish according to the size selectivity. The complete configuration of the model is presented in Table 5.

Other settings include e.g. fecundity. We have assumed that egg output is proportional to spawning biomass, which is reasonable as there is no clear relationship between relative fecundity (number of eggs per 1 g of female) and length (Lappalainen et al. 2003).

Table 5. Settings of the pikeperch model. The table includes number of estimated parameters, the initial values (from which the numerical optimization is started), the intervals allowed for the parameters, the value estimated by the model and its standard deviation. Parameters in bold are fixed and not estimated by the model.

Parameter	Number estimated	Initial value	Bounds (low,high)	Estimated value	Standard deviation
Natural mortality (ages: 0.5, 1.5, 5, 15)		**0.916, 0.529, 0.276, 0.188**			
Recruitment					
Ln(R_0)	1	9	(1, 30)	8.29	2
Steepness (h)		**0.93**			
Recruitment variability (σ_R)		**0.5**			
Ln (Recruitment deviation): 1966 - 2018	52				
Recruitment autocorrelation		**0**			
Initial catches		Mean of catches in 1966-1968			
Initial F trapnet fleet	1	0.009	(0.001, 1)	0.185	0.1

Initial F gillnet fleet	1	0.009	(0.001, 1)	0.214	0.1
Length selectivity					
Trap net fleet 1966-2000					
peak		**40**			
top_logit		**-15**			
ascend_se		**-4.36**			
descend_se		**20**			
2001-2021					
peak		**45**			
top_logit		**-15**			
ascend_se		**-4.36**			
descend_se		**20**			
Gillnet fleet 1966-2000					
peak		**60.09**			
top_logit		**-15**			
ascend_se		**6.01**			
descend_se		**25**			
2001-2021					
peak	1	56.02	(4, 74.5)	54.53	99
top_logit		**-15**			
ascend_se	1	4.297	(-20, 15)	3.98	5
descend_se		**25**			
Historical gillnet survey					
peak	1	15.86	(4, 60)	41.22	99
top_logit		**-15**			
ascend_se	1	3.85	(-15, 8)	4.6	5
descend_se		**20**			
Recent gillnet survey					
peak	1	15.86	(4, 60)	37.17	99
top_logit		**-15**			
ascend_se	1	3.85	(-15, 8)	7.39	5
descend_se		**20**			
Catchability					
Historical survey					
Ln(Q) – catchability	1	-2.48	(-25, 25)	1.57	1
Extra variability added to input standard deviation		**0.1**			
Recent survey					
Ln(Q) – catchability	1	-2.48	(-25, 25)	-0.58	1

Trapnet fleet					
Ln(Q) – catchability	1	-2.48	(-25, 25)	-5.16	1
Extra variability added to input standard deviation		**0.1**			
Gillnet fleet					
Ln(Q) – catchability	1	-2.48	(-25, 25)	-2.75	1
Extra variability added to input standard deviation		**0.1**			

5.4. Model diagnostics

The selectivity curves all look reasonable with historical and recent surveys selecting smaller pikeperch than the gears in the commercial fleets (Figure 27, Figure 28). The trap net fleet has the ability to actively select which individuals to land and can release fish under the minimum size. Released fish likely has a high survival rate (Nyberg et al. 1996; Dannewitz et al. 2010) and we therefore applied a threshold selectivity without discard. The gillnet fleet on the other hand can catch under-sized fish and has an estimated selectivity similar to the historical survey nets, but selecting larger fish (parallel yellow and red lines on Figure 27).

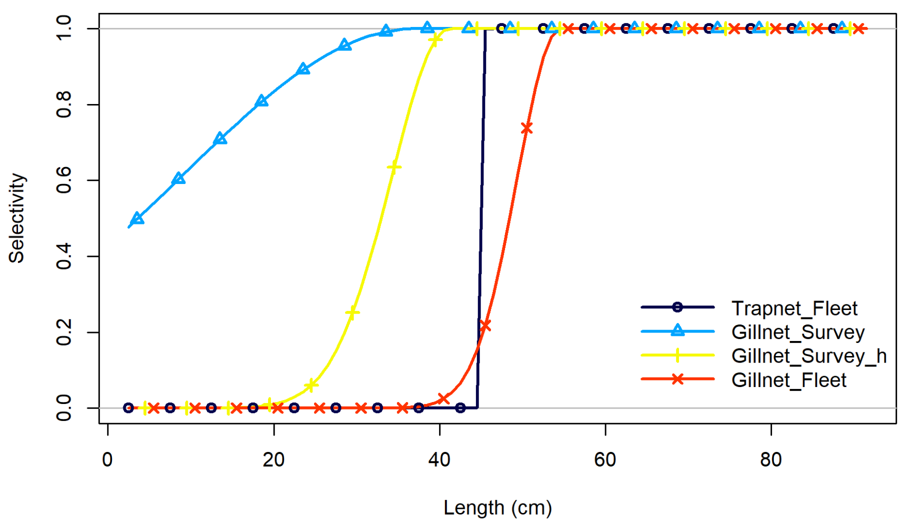


Figure 27. Length-based selectivity in 2021 of commercial fishery using trap nets and gillnets, historical survey (Gillnet_survey_h) and recent survey (Gillnet survey).

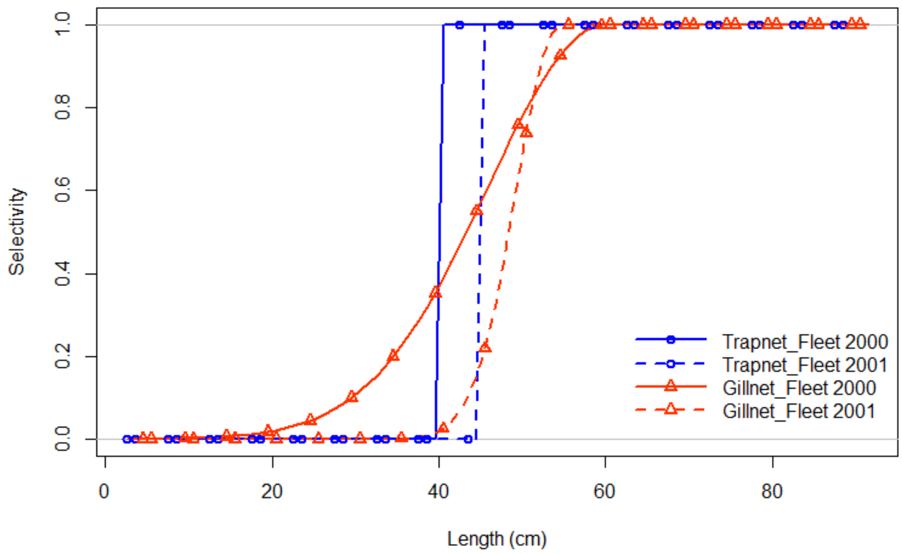


Figure 28. Length-based selectivity of commercial fishery using trap nets and gillnets in 1966-2000 (2000) and 2001-2021 (2001).

The model fitted length distributions quite well to the trap net fleet, gillnet fleet and the historical gillnet survey, while the recent gillnet survey had a poorer fit (Figure 29).

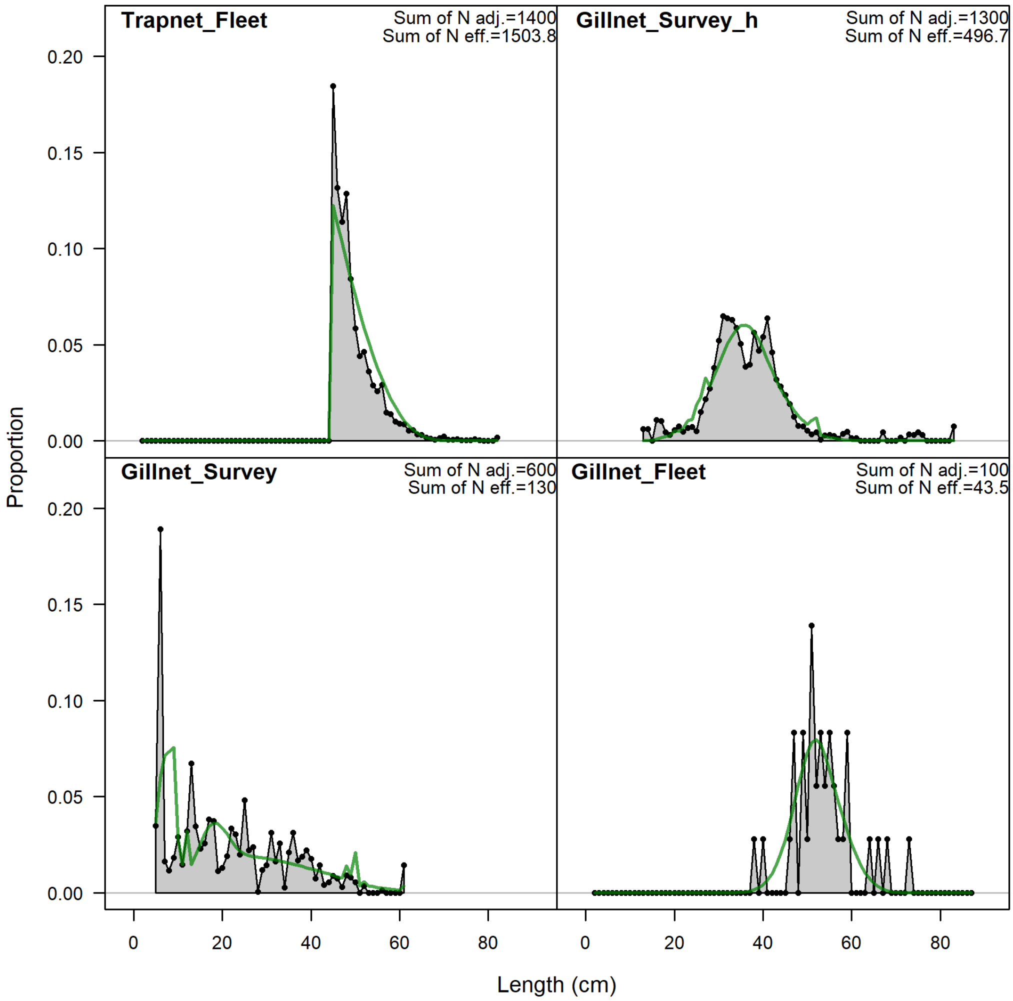


Figure 29. Model fit (line) to observed length distribution data (grey polygon).

Overall, the model provided quite good fit to the trends in the historical and recent survey indices (Figure 30, Figure 31). As for the fit to the fisheries indices (which ended 2017) it was a better fit for the trap net fleet compared to the gillnet fleet (Figure 32, Figure 33).

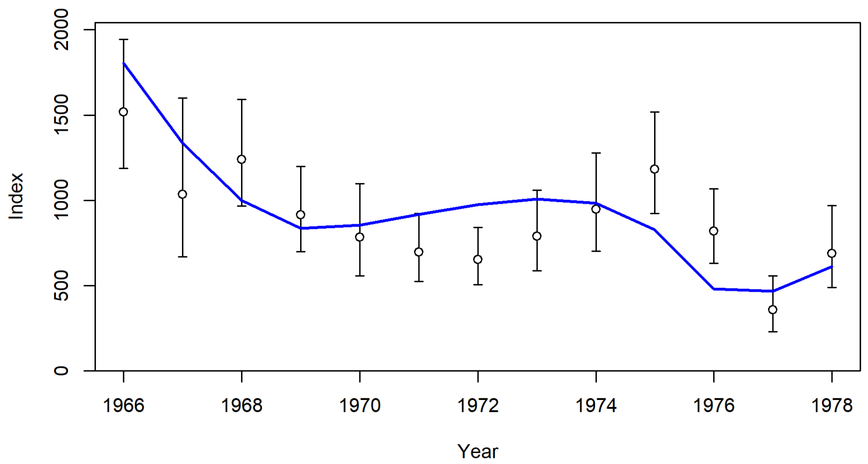


Figure 30. Model fit (line) to abundance indices from historical gillnet survey (whiskers indicate CV).

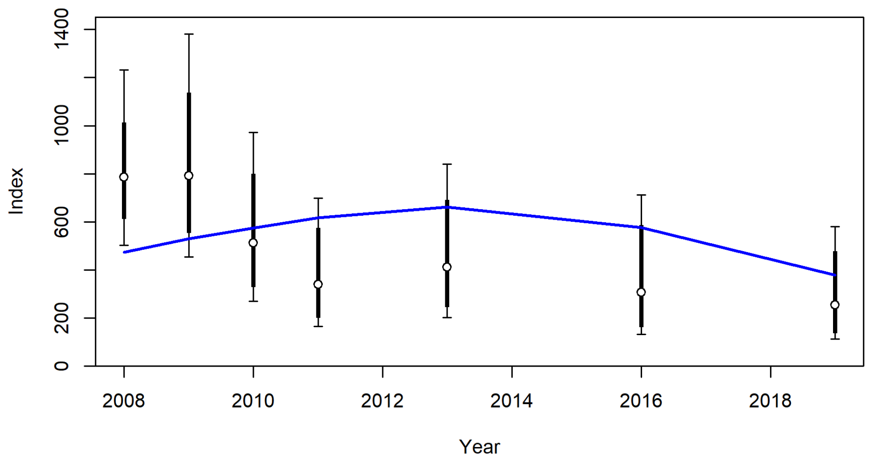


Figure 31. Model fit (line) to abundance indices from recent gillnet survey (whiskers indicate CV).

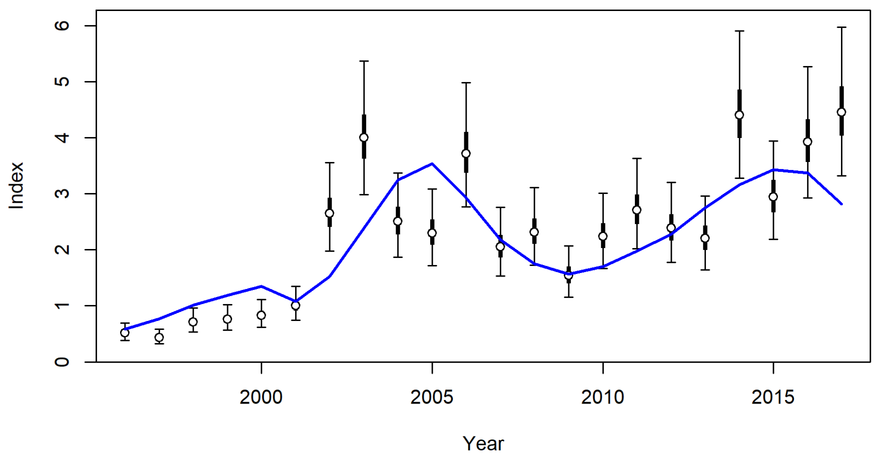


Figure 32. Model fit (line) to stock indices from the trap net fleet (whiskers indicate CV).

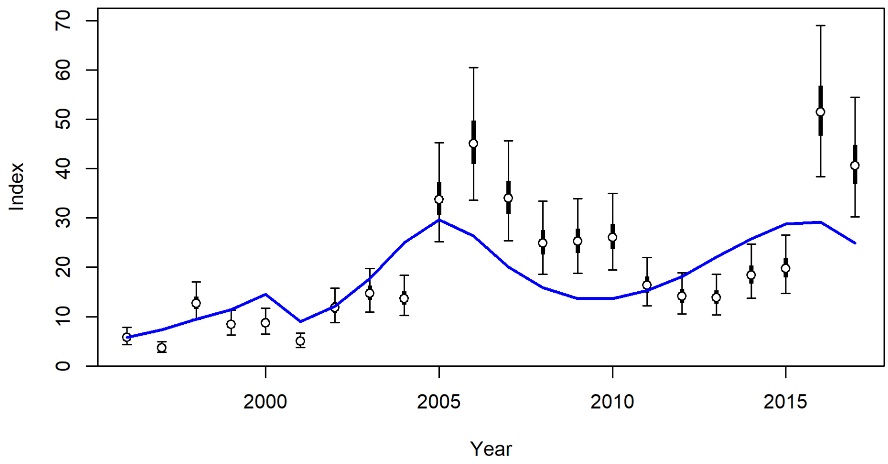


Figure 33. Model fit (line) to stock indices from the gillnet fleet (whiskers indicate CV).

5.4.1. Retrospective analyses

Retrospective analyses are used to evaluate the reliability of parameters and reference point estimates and to show potential systematic bias in the model estimation (Hurtado-Ferro et al. 2015; Szuwalski et al. 2018). It starts with fitting a stock assessment model to the full dataset and subsequently fitting the model to a gradually decreased dataset where the data are removed for the most recent year one by one.

The retrospective analysis was run for the last 5 years of the assessment and did not indicate that the model have instability (Figure 34).

In addition, Mohn's rho index (a common measure for retrospective patterns (Hurtado-Ferro et al. 2015)) was estimated for SSB. As a rule of thumb, long-lived species have been proposed to have rho-values within a range of -0.15 - 0.20, while for short-lived species within -0.22 - 0.30 (Hurtado-Ferro et al. 2015). Values outside those ranges may indicate concerns about the retrospective patterns. The estimated variant of the Mohn´s rho indices were within the recommended range for values of SSB (0.009).

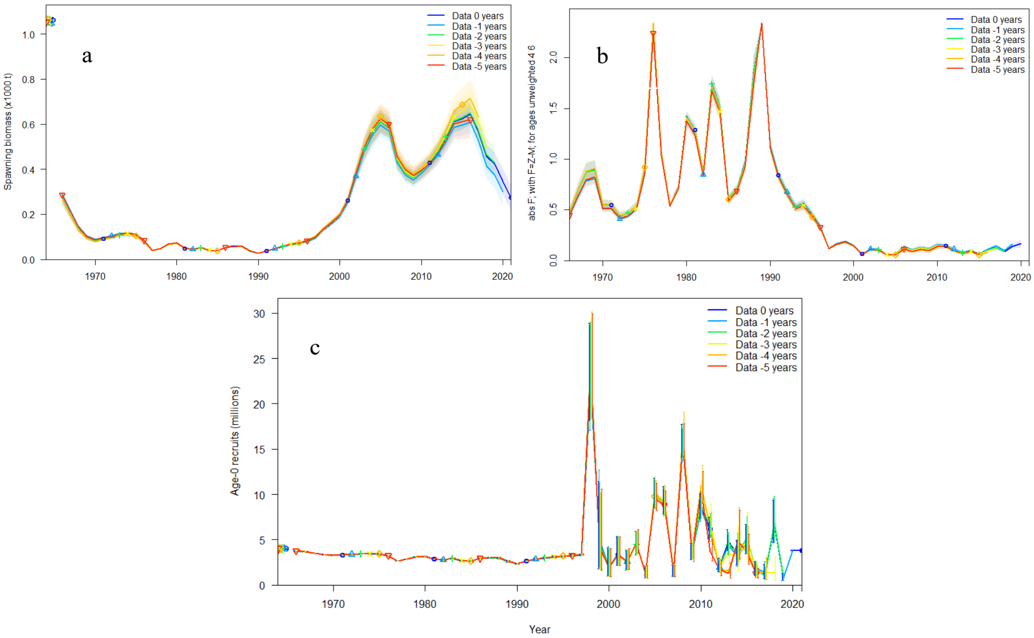


Figure 34. Retrospective analyses of the model output: SSB (a), F4-6 (b) and recruitment (c).

5.4.2. Trends in SSB, F and R of the reference model

The spawning stock biomass (SSB; Spawning biomass on Figure 35) declined in the beginning of the time series, increased during the 1990s and then fluctuated and declined drastically from 2016. Fishing mortality (F) had a decreasing trend with some fluctuations in the beginning of the time series, but most pronouncedly decreased from 1990. There is slight increase from 2018. Recruitment (R) has been fluctuating after 1998, but rather stable prior to it due to limited age composition data. In 1998 and 2008 strong year classes appeared (recruitment in Figure 35).

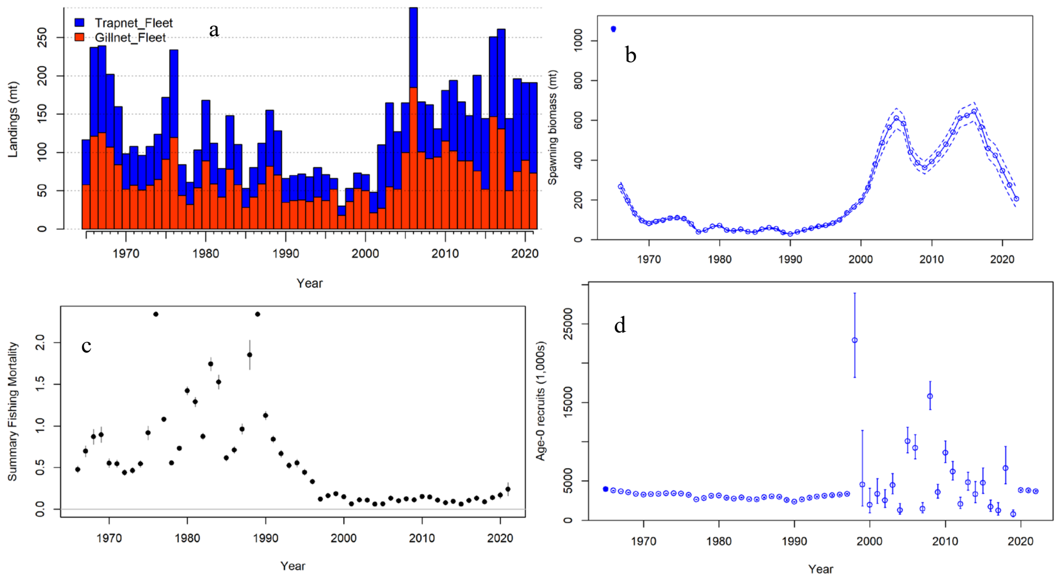


Figure 35. Summary of reported landings (a) and the output of the assessment model (b-d). SSB (b), F4-6 (c) and recruitment (d) are shown with 95% confidence intervals. Landings by fleet and SSB are in tonnes, recruitment is in thousands of individuals.

5.4.3. Alternative model formulations

To assess the model described above (called reference model), and to test potential changes that we thought could improve the general fit of the model, we investigated two alternative model formulations:

1. To check the effect on model of fixing selectivity parameters, instead of using predefined fixed values of the peak and ascend parameters of the trapnet fleet in 2001-2021, we allowed the parameters to be estimated in the model

2. To check whether abundance indices would fit better if the model could focus on them instead of trying to fit all the data sets we downweighted the length distributions and ALKs by using Dirichlet-multinomial error and emphasis factors (lambdas)

The output of the reference model and model alternative 1 are nearly identical, but there are some differences in model alternative 2 (Figure 36, Figure A9). Model alternative 2 suggests less of a decline and higher SSB in recent years compared to the reference model, and also has wider confidence intervals around the SSB estimate (Figure 36, subplot a). Related to that, model 2 also suggest that fisheries mortality (F4-6) has been lower towards the end of the time-frame (Figure 36, subplot b). Recruitment estimates from model 2 (Figure 36, subplot c) were different in some years (2000-2007, 2010-2015), but similar in other years, including the most recent years where SSB and F differ (1998, 2008, 2016-2021).

The reference model and model alternative 1 fit nearly identically to survey indices and mean length (Figure 37 and Figure 38), but model alternative 2 fit a little worse to recent survey indices and a little better to mean length from recent survey data (Figure 37 and Figure 38). This is counterintuitive since the reason we down weighted length distribution and ALKs was to improve fit to indices.

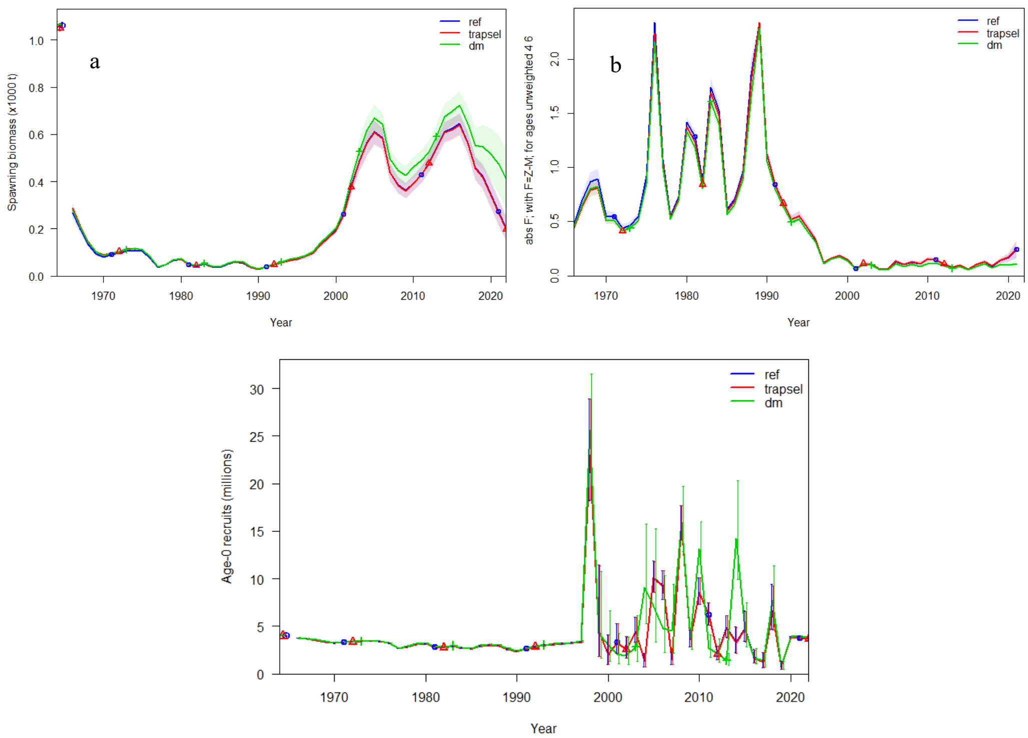


Figure 36. Comparison of model output between reference model (ref), model alternative 1 (trapsel) and model alternative 2 (dm): SSB (a), F4-6 (b) and recruitment (c) shown with 95% confidence intervals. SSB are in tonnes, recruitment is in thousands of individuals

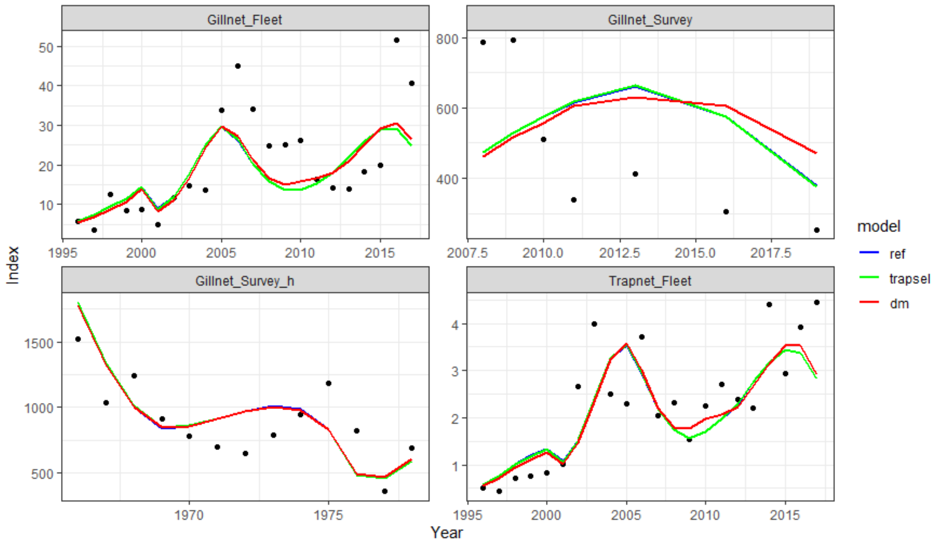


Figure 37 Comparison of model fit to biomass indices between reference model (ref), model alternative 1 (trapsel) and model alternative 2 (dm).

Figure 38 Comparison of model fit to mean length between reference model (ref), model alternative 1 (trapsel) and model alternative 2 (dm)

5.5. Conclusions and recommendations

Building a reference model based on limited data sources was as expected a challenging task. The model estimates mean age- and length development quite well (Appendix Fig A4-A8) and the fit to the length distributions and indices (Figure 29, Figure 30, Figure 31, Figure 32) is generally also good (except for gillnet fleet indices; Figure 33). However, it should be noted that the recent gillnet survey (Figure 31) and the gillnet fleet index (Figure 33) that overlap in time indicate different temporal trends. The stock assessment model can therefore not be expected to fit both indices, and we expect a poorer fit to one of the two indices. Both the reference and alternative model formulations provide a better fit to the fleet index than the survey index, which could be an effect of the fleet containing more years of data and therefore carry more information content (from the models perspective).

Although model fit is not optimal, the model indicates a declining trend of the spawning stock biomass (Figure 35b), which may indicate a deteriorating status of the stock. However, the alternative model formulation 2 suggest this decline to be less sever (although still a decline, Figure 36). The model currently includes data up to and including 2021, and new model runs with data from 2022 should provide additional information that could improve the models.

As management objectives and reference values have not been agreed upon, a more formal stock assessment cannot be made. **We therefore recommended that managers initiate a discussion on management goals and reference levels for this stock. The management objectives should consider mortality, biomass as well as size-based targets.** Several outputs from the model can be used to assess management objectives, the most important ones are spawning stock biomass, recruitment and fisheries mortality.

Additional actions that can be taken to further improve the model:

- Better spatial coverage – that includes both length and age data from commercial catches. The model currently contains information from only one fisher and an increased coverage should include more fishers and both fleets, i.e. trapnet and gillnet, to ensure that the commercial catch composition is accurately reflected in the model.
- Revisit historical and recent survey abundance indices and assess various modelling techniques to better describe the stock development, e.g. by including not only site and gear, but also temperature and other environmental factors (since catches in survey can be influenced by e.g. temperature (Naddafi et al. 2022)).
- Update archived age data from survey (Figure A5, years 2013 and 2016).
- Use the models described in this status report and continue diagnostics, particularly hindcasting, retrospective and runs test of the residuals to select the "best case" model for advice.

Complementary studies:

- The current catch index from commercial catches ended in 2017. With the new digital reporting tool ("EFR"), we anticipate a development of a much needed stock index.
- Acoustic telemetry could give insights into migration patterns within the lake as well as a measure of total mortality. This would provide input to model assumptions regarding representatively of different sites and indices.
- Further investigations of selectivity in commercial gears, e.g. by testing the different mesh sizes used, especially 100 mm and 120 mm mesh sizes (50 and 60 mm) but potentially also larger meshes.
- Compare old and current monitoring sites using the current gear (Bkust9+2), in order to provide input data for an adjustment of the survey indices.
- Assess landings in recreational catches compared to commercial catches.

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Appendix

Additional information for the pikeperch assessment model including estimates of parameters outside the model and additional diagnostic plots.

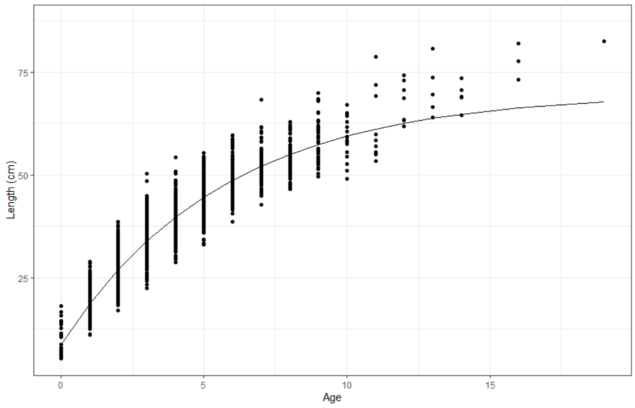

Figure A1. Estimated growth curve (line) fitted to length at age (dots) pooled from 2008-2021 data from survey and fisheries.

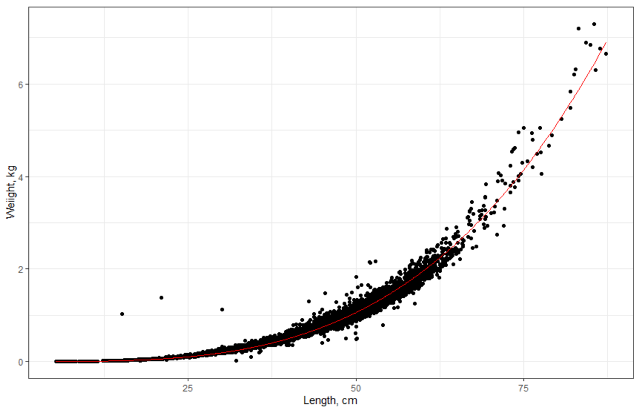

Figure A2. Estimated length-weight relationship (line) fitted to weight at length (dots) pooled from 2008-2021 data from survey and fisheries.

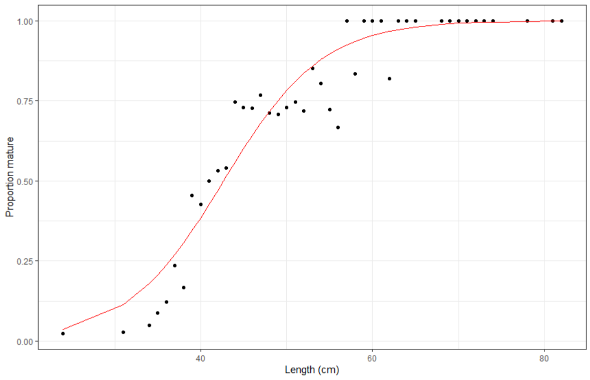

Figure A3. Estimated maturity ogive (line) fitted to proportion mature at length (dots) pooled from 2008-2021 data from survey and fisheries.

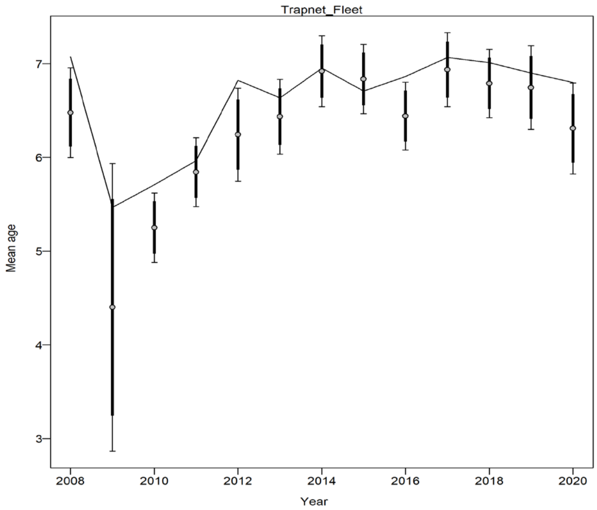

Figure A4. Comparison of estimated mean age by model (line) and observed from commercial fisheries (dots with whiskers).

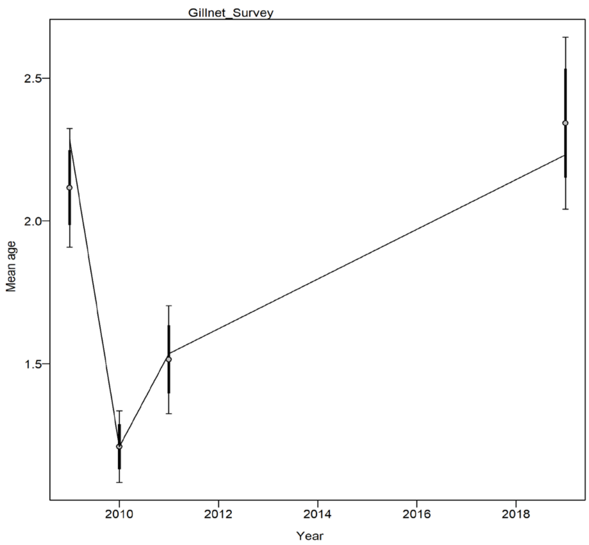

Figure A5. Comparison of estimated mean age by model (line) and observed from recent survey (dots with whiskers).

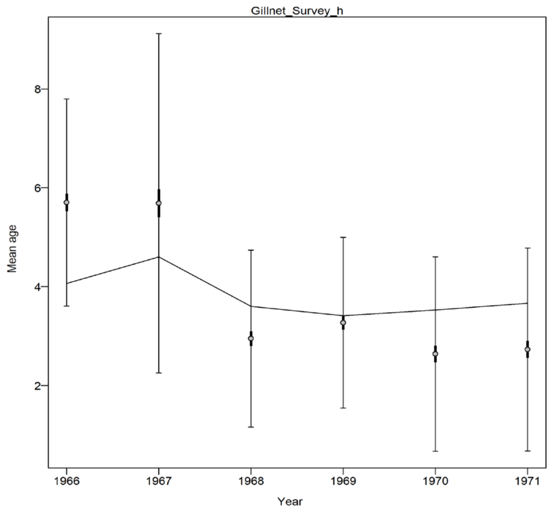

Figure A6. Comparison of estimated mean age by model (line) and observed from historical survey (dots with whiskers).

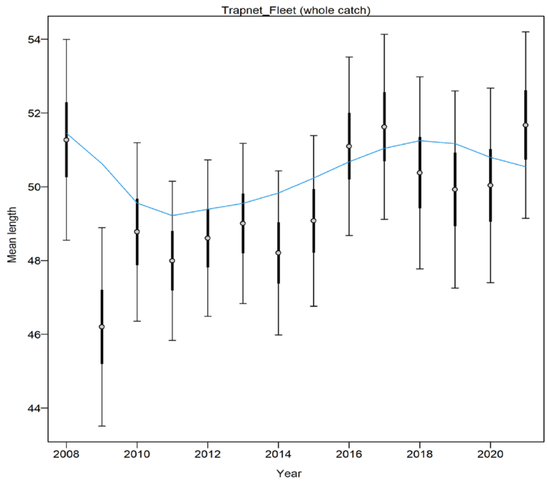

Figure A7. Comparison of estimated mean length by model (line) and observed from commercial fisheries (dots with whiskers).

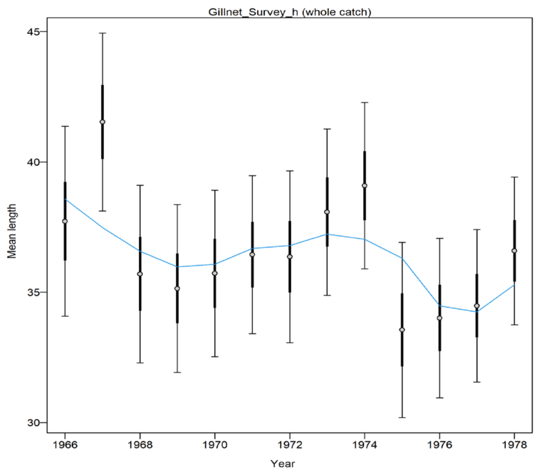

Figure A8. Comparison of mean length estimated by model (line) and observed from historical survey (dots with whiskers).

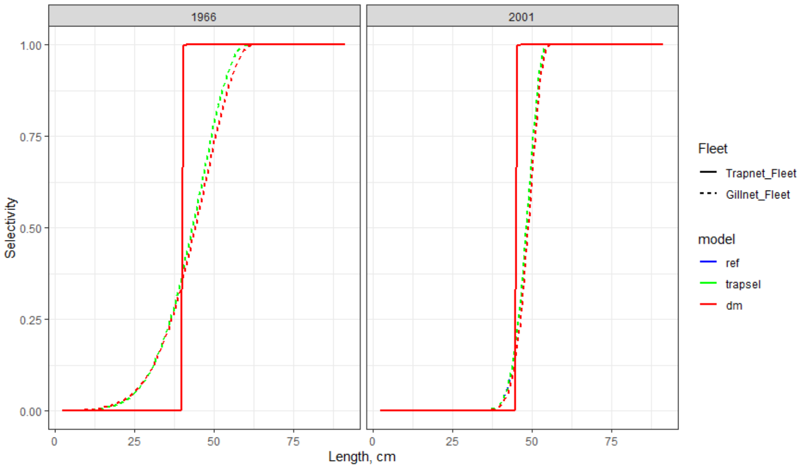

Figure A9. Comparison of fleet selectivity between reference model (ref), model alternative 1 (trapsel) and model alternative 2 (dm)