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Contents

Summary

Shellfish are important to Scottish fisheries; a total of 61 544 tonnes of shellfish with a value of £150 million were landed by Scottish vessels in 2014. However, for many shellfish stocks only limited biological data are available and the lack of appropriate monitoring hampers stock assessment and the provision of scientific advice. The data limitation is due to the lack of ageing methods for crustaceans. For sedentary stocks exploited by localised fisheries, data and assessment at the appropriate spatial scale are often not available.

In this report, we review the use of indicators in the assessment and advisory process and management of data limited stocks, both fish and shellfish, worldwide. For the purposes of the review, we define an 'indicator' as quantifiable information that acts as a proxy for, or can be related to the state of the stock (spawning stock biomass, demographic properties or recruitment) and anthropogenic pressure (fishing mortality). Measured (or derived) regularly, indicators are able to show changes in the state of the population or system and can give information on stock status and fishing mortality and support a data-limited stock assessment. Used alone or more often in combination, indicators can provide the means to assess progress towards one or more management objectives. We describe the derivation and theoretical basis of a variety of indicators which can be calculated from commercial fishery data, fishery-independent survey data and biological data, and the ways in which indicators are used in fisheries management. We give details on methods for selecting, evaluating and combining indicators, for developing management strategies, determining reference points and for the testing of harvest control rules (HCRs). We then consider examples of shellfish stocks worldwide for which an indicator-based approach has been applied. Finally, we consider candidate indicators for Scottish shellfish stocks and fisheries and discuss their development in the context of existing and potential future data collection programmes.

The work is part of research being carried out by Marine Scotland Science under a ROAME project SU0100, which is aimed at developing guidelines for the selection and application of appropriate indicators for shellfish fishery management. This work will also inform the development of indicators of good environmental status (GES) for shellfish as required under the Marine Strategy Framework Directive (MSFD) (EC, 2008b).

1 Introduction

1.1 Background

Scotland has a highly diverse shellfish fishing industry which includes trawling for *Nephrops*, creel fisheries for a variety of crustacean species, scallop dredging and hand gathering for a range of mollusc species. In 2014 shellfish landings by Scotland-based vessels were 61 544 tonnes with a value of £150 million (SG, 2015). Shellfish are key species for the Scottish inshore fleet, and there are significant fisheries for both Norway lobster (*Nephrops norvegicus*) and brown crab (*Cancer pagurus*) offshore. Although difficult to quantify precisely, the economic benefits accruing from these fisheries play an important role supporting rural communities.

Scottish shellfish fisheries are primarily regulated through EU technical conservation rules (EC, 1998), which include measures such as minimum landing sizes (MLS), and through Orders made under the Sea Fisheries (Shellfish & Conservation) Act 1967 and the Inshore Fishing (Scotland) Act 1984. In addition, vessels must have a licence with a shellfish entitlement. There are no national or international quotas on landings of Scottish shellfish stocks, except for *Nephrops,* which are managed through total allowable catches (TAC) and quotas set through EU legislation.

The Scottish Government's review of inshore fisheries published in 2005 (SE, 2005) led to the establishment of Scotland's Inshore Fisheries Groups (IFGs), non-statutory stakeholder bodies with the overall aim of improving the management and conservation of Scotland's inshore fisheries (out to six nautical miles). As part of their remit, the IFGs have developed management plans and some are actively involved in proposing and implementing local measures for sustainable exploitation of commercial stocks.

The provision of fisheries management advice typically relies on the comparison of estimates of fishing mortality and spawning stock biomass (current stock status) to a series of thresholds known as reference points. Usually, the estimates are derived from a stock assessment model based on both surveys and fisheries-dependent data. Marine Scotland Science (MSS) conducts regular stock assessments for *Nephrops* and a number of the non-quota shellfish species of interest to the IFGs, including stocks of the scallop, brown and velvet crab (*Necora puber*) and European lobster (*Homarus gammarus*) around Scotland (Howell *et al.*, 2006; Dobby *et al.*, 2012; Mesquita *et al.*, 2016). Available data include fisheries-independent data from surveys as well as fisheries data and biological data from commercial sampling.

However, in some areas there are insufficient data to conduct assessments and in others, the assessments are not carried out sufficiently frequently or at the appropriate spatial scale to enable monitoring of the effectiveness of specific local management measures. Difficulties associated with shellfish stock assessment are not limited to Scottish stocks. For many shellfish it is not possible to apply standard stock assessment methods to evaluate stock status. The application of age-structured assessment methods to crustacean stocks is problematic, because age determination is generally not possible. It is a feature of crustacean biology that in order to grow individuals moult to renew their exoskeleton. Due to a lack of persistent hard structures necessary for age-reading, age-length keys are not available for crustacean species. Length-structured models may be more appropriate, but these are often highly complex and require considerable data input (Smith and Addison, 2003; ICES SGASAM, 2005; Punt *et al.*, 2013a). Sedentary mollusc species like scallops, for which age-structured data can be provided, are often exploited by very localised fisheries, and it can be difficult to obtain data and to conduct assessments at the appropriate spatial scale. The development of indicators of stock status, which can be derived from analyses of data gathered through existing monitoring programmes or through new schemes run by the IFGs themselves, is therefore essential.

For data-limited stocks (not solely shellfish) the use of indicators in the provision of fisheries management advice is becoming more widely accepted (Trenkel *et al.*, 2007; Ye *et al.*, 2011; Babcock *et al.*, 2013). In some cases indicators are used to provide qualitative (or 'soft') management advice based on analysis of trends, but there are others in which indicator-based frameworks are being used in the provision of quantitative advice (ICES, 2012). Such quantitative indicators could potentially be incorporated into the development of formal (pre-agreed) decision-rule-based management plans.

1.2 Types of Indicators

Indicators which are calculated directly from raw data (e.g. Catch Per Unit Effort (CPUE) from either survey or commercial data) are known as 'empirical indicators' whilst indicators which are derived from a range of data and parameters (e.g. fishing mortality estimates from a virtual population analysis) are known as 'estimated indicators' (Scandol, 2005). There are different types of indicators available such as stock status indicators, indicators of effects of fishing, indicators of economic and social outcomes, and indicators of regulatory compliance.

Within the European Union, there are frameworks to ensure good environmental status and support scientific advice with regard to the Common Fisheries Policy (EC, 2002). The EU Data Collection Framework (DCF) regulates the sampling and collection of commercial fisheries data. This includes the collection of economic variables, and biological variables of catch samples, i.e. métier-related (catch composition, discards) as well as stock-related variables (individual ages, length, weight, sex, fecundity). These data can be used to calculate a number of indicators which help to evaluate the stock status and the effect of the fisheries on the marine ecosystem (EC, 2008a), i.e. conservation status, proportion of large individuals, mean maximum length of species, size at maturation of exploited species, distribution and aggregation of fishing activities, areas not impacted by mobile bottom gears, discarding rates of exploited species, and fuel efficiency of capture.

The EU Marine Strategy Framework Directive (MSFD) (EC, 2008b) requires that member states achieve good environmental status (GES) of marine waters by 2020. MSFD Descriptor 3 requires that 'Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.' The criteria for MSFD Descriptor 3, relate to fishing mortality, reproductive capacity of stock and its age/size distribution. These are connected to indicators which may reveal fluctuations in the stock status (EC, 2010).

Rice and Rivard (2007) considered two different roles of indicators in assessments and advice. Indicators can have an 'audit' function, reporting on the effectiveness of previous management in meeting particular biological or economic objectives (Smith *et al.*, 1999). This contrasts with their 'control' function, when they are used to guide decision-making and policy-setting for future stock management based on the difference between the current value of the indicator and biologically-based reference points (Garcia and Staples, 2000). The choice of appropriate indicators is conditioned by the function they need to fulfil.

Biological indicators that characterise a single stock are known as 'population indicators' whilst the term 'community indicator' is used when they apply to a group of species or stocks (Rochet *et al.*, 2005). This review is primarily concerned with the use of empirical 'population indicators' in the stock assessment and management process rather than in the description of the status of an ecosystem. Single indicators can be used or multiple indicators can be evaluated sequentially, collectively or hierarchically (Dowling *et al.*, 2015).

1.3 Reference Points and Harvest Control Rules (HCRs)

In order to be able to evaluate how the observed value of a particular indicator relates to management objectives, appropriate reference points need to be defined. There are two kinds of reference points. Target reference points indicate the desirable outcome of successful fisheries management. To fulfil management objectives, actions usually aim to maintain a variable at or near a target reference point and prevent it from, on average, exceeding or falling below the threshold. In contrast, limit reference points mark the boundaries between acceptable and unacceptable outcomes of management. Following the precautionary approach, values beyond these limit reference points indicate damage to the resource and the requirement of action (Caddy and Mahon, 1995; Caddy, 2002). Further general discussion of the choice and derivation of reference points can be found in Sissenwine and Shepherd (1987), Goodyear (1993) and Mace (1994).

Reference points are often based on population dynamic models or growth models, taking into account available biological knowledge and life histories of a stock. Management frameworks typically include reference points for fishing mortality and/or spawning stock biomass based on Maximum Sustainable Yield (MSY). MSY describes the largest average catch, that can continuously be taken from a stock without causing stock decline and has been a widely accepted fisheries management objective for many years (United Nations Convention on the Law of the Sea (UNCLOS, 1982)), and adopted in many fisheries management systems. The fishing mortality that produces MSY, F_{MSY}, (or arbitrary conservative proxies) has been used as a limit fishing mortality to keep stocks within safe biological limits. Reference points are set to avoid recruitment failure and ideally are derived on the basis of a stock-recruitment relationships and yield-per-recruit (YPR) models (Gabriel *et al.*, 1989; Mace and Sissenwine, 1993; Garcia, 1996; Myers and Mertz, 1998), which are often estimated as part of the stock assessment process.

For data-limited stocks, however, the available data are not adequate for analytical stock assessments or the calculation of reference points based on stock biomass and fishing mortality. A number of other methods, including the Catch-MSY procedure (Froese and Binohlan, 2000; ICES WKLIFE, 2012a; Martell and Froese, 2013; ICES WKLIFE, 2015), have been developed to estimate MSY reference points from a catch time series in such cases. The Catch-MSY method is based on a Schaefer production model and provides estimates of the stock biomass, MSY and F_{MSY}, which best explain the observed catches, given a rough estimate of the depletion level. Methods based on historical time series of an indicator rather than population models have been used mostly to supplement the traditional basis for management advice (Cadrin *et al.*, 2004). Non-analytical approaches may aid in development of an empirical reference point system (Punt *et al.*, 2001a; Hilborn and Stokes, 2010). In this type of approach, reference points are chosen on the basis of a combination of expert opinion and comparison of current values to data from earlier periods in the fishery. However, as indicator values include error, precautionary reference points would need to be used to ensure a high probability that the limit is avoided (FAO, 1998). For a biomass indicator series, such as CPUE from survey or commercial fisheries, the lowest observed indicator value or a percentage of the historical mean or maximum observed index have been suggested as limit reference points (Smith *et al.*, 2012).

Reference points for length-based indicators can be based on published life history characteristics and parameters of the von Bertalanffy growth model, such as length at 50% maturity L_{mat}, asymptotic length L_∞, or optimal harvest length L_{opt} (length class at maximum biomass of unexploited cohort, about 2/3 L_∞) (ICES WKLIFE, 2012b; ICES WKLIFE, 2014). L_∞ can also be approximated by observed maximum length or percentiles of length distributions of historic time series for a stock. Analyses of life history characteristics help to evaluate whether a stock is prone to overexploitation or can compensate for fishing pressure (Cope, 2006). Slow-growing, late maturing, K-selected species, which produce relatively few offspring and are regulated by carrying capacity, tend to be more vulnerable to fishing (Jennings *et al.*, 1999) than r-selected species which are smaller and early maturing. The reference point L_{mat} could be used to test whether enough (K-selected) fish survive to maturity and spawn before becoming vulnerable to fishing gear (Caddy and Mahon, 1995).

Once indicators and reference points have been agreed, HCRs can be developed, evaluated and applied. HCRs help to automate management decisions. An HCR is used as a feedback control between the current state of the stocks, as reflected by respective indicators and reference points, and the advice on future catches. In an HCR scenario, indicators take on a control function, where a discrepancy between the current indicator value and reference point may trigger a change in the fishing regime (Rice and Rivard, 2007).

Simple HCRs include, for example, (i) constant catch, (ii) constant escapement, and (iii) constant fishing mortality (Lande *et al.*, 1995; Deroba and Bence, 2008). The constant catch rule ignores stock status and does not require an annual stock assessment. Under this harvesting regime in a variable environment, stock collapse may occur. The constant escapement rule aims to leave a constant stock abundance surviving fishing mortality over a spawning cycle, such that stock abundance remains near a target, and thereby avoids collapse. At constant fishing mortality, the proportion of the stock caught remains constant (assuming that other sources of mortality also do not change). Keeping fishing mortality constant was found to perform well for fish stocks with various life-history types (salmon, cod, herring-like), when accounting for auto-correlated environmental effects and recruitment uncertainty (Walters and Parma, 1996; Vasconcellos, 2003; Brunel *et al.*, 2010). HCRs can be modified using thresholds or multiple thresholds to avoid excess harvesting (Enberg, 2005; Zhang *et al.*, 2011). However, HCRs based on either constant escapement or constant fishing mortality require information on stock abundance.

For data-limited stocks, for which stock abundance is unknown, alternative simple HCRs can be developed based on catch time series (Dichmont and Brown, 2010; ICES, 2012). Any summary statistic of a historic catch series can be used to set future catches, including the mean or median historical catch or a percentage thereof, the mean catches of the last three years, the median of the previous 10 years, or the third highest catch in the last 10 years (Carruthers *et al.*, 2014). HCRs can be further developed by the adjustment of previous year's catch with an abundance/biomass indicator trend, a length-based indicator, or in relation to uncertainty (ICES WKLIFE, 2013). Simulation tests of these HCRs showed good performance in determining sustainable catches when a length-based indicator was included to adjust future catches (Carruthers *et al.*, 2014).

The Restrepo approach for data-limited stocks (Restrepo *et al.*, 1998) is a method for deriving an appropriate TAC on the basis of historical catches and a qualitative judgement of the stock status. Future catch is calculated as the product of the average catch over a reference period with relatively constant catches and stable abundance, and a multiplier corresponding to the stock status category (between 0.25 for overexploited stocks and 1.0 for stocks above B_{msy}). In this approach, future catch will never exceed the average of the reference period and remains constant as long as the stock status category is the same.

The Methot Table Conceptual Framework, which is similar to the Restrepo method, follows a qualitative approach in which stocks are categorised according to both the exploitation and productivity level. The future catch is then determined from the recent average catch over a stable period and the respective scalar multiplier according to stock vulnerability (Berkson *et al.*, 2011). The approach of the Only Reliable Catch Stock (ORCS) Working Group combines the Restrepo and Methot methods and works in three steps (Berkson *et al.*, 2011): (i) a scoring procedure is used to categorise stocks by exploitation level, (ii) a scalar multiplier according to exploitation level together with the recent average catch determines the overfishing limit, and (iii) the acceptable catch is then a proportion of the overfishing limit depending on the productivity status. In these methods, the acceptable catch is constant only so long as the perceived exploitation and productivity level remains unchanged.

In the US, following recent changes to the Magnuson-Stevens Fisheries Conservation and Management Act (NOAA, 2010), fishery managers must implement annual catch limits for all fisheries, including those for data-poor stocks. As part of this process, two methods for advising on sustainable catch limits in data-poor situations were explored for various demersal fish stocks: the Depletion-Corrected Average Catch (DCAC) (MacCall, 2009) and the Depletion-Based Stock Reduction Analysis (DB-SRA) (Dick and MacCall, 2011). Both methods use catch histories and allow for changing population abundance over time. They were initially developed for a short-term management. The DCAC method combines information on average catch with estimates of natural mortality and expert opinion on the depletion level of the stock to estimate sustainable harvest (MacCall, 2009; Dick and MacCall, 2011). In this context, the historical catches are assumed to consist of two components, one part of the catch which is sustainable and the remainder which would cause the stock abundance to decline. To maintain a stock near the levels experienced during the period of the catch time series, the average historical catch is 'corrected' providing an estimate of sustainable yield. However, if the stock has experienced a recent severe depletion or high rates of natural mortality this method may not be suitable to estimate sustainable yield.

As an extension of the DCAC method, the Depletion-Based Stock Reduction Analysis (DB-SRA) requires additional information on maturation and a complete catch time series (Dick and MacCall, 2011). Dick and MacCall (2011) compared the DB-SRA results of 31 data-rich demersal fish stocks with their full assessment results showing that the method can provide sustainable yield estimates. Carruthers *et al.* (2014) discuss how results from the DCAC and DB-SRA can be used in the development of catch-based harvest control rules. Both DCAC and DB-SRA were found to be sensitive to the assumption of stock depletion (Wetzel and Punt, 2011). For the related catch curve stock reduction analysis (CC-SRA) no prior on stock depletion is necessary, and the method was found to be unbiased at low/moderate recruitment variability (Thorson and Cope, 2015).

Wiedenmann *et al.* (2013) simulation-tested various HCRs, which use the Restrepo, ORCS, DCAC or DB-SRA method for three different life history scenarios ('slow', ‘medium’ and ‘fast’ life histories) combined with three different exploitation histories (under, fully and over-exploited). They found that no single approach performed best in setting sustainable catch limits in most scenarios of stock life history and exploitation level, such that misspecification and approach selection can have strong effects on the outcome. Control rules, for example the Restrepo approach, that contain broad categories of stock levels and apply to various life histories are advantageous. Arnold and Heppell (2015) compared the DCAC to the DB-SRA method for canary Rockfish (*Sebastes pinniger*). They showed that while the DCAC method in general is more conservative, the DB-SRA is more precautionary at high depletion levels. Berkson *et al.* (2011) suggested using DB-SRA whenever possible followed by DCAC, and then ORCS if the other two methods are not appropriate.

2 Indicators of Abundance

2.1 Abundance Indicators from Surveys

Trawl or dredge surveys typically provide estimates of abundance in terms of catch rates (CPUE), in either number or biomass per hour or per distance towed or by area swept. Using CPUE data as an indicator relies on the critical assumption that CPUE is proportional to abundance, which requires catchability to remain constant. In surveys, standardised sampling design and equipment can help to maintain constant catchability. However, catchability may be influenced by individual length and season. In addition, changes in spatial distribution, movement and migration of stocks, may affect catchability in ways which may be independent of stock abundance (Hilborn and Walters, 1992; Rose and Kulka, 1999). Since CPUE often shows high inter-annual variability, the index is often transformed and log(CPUE) used as the indicator for abundance (with CPUE in kg h^{-1} or individuals km^{-2}, for example). Underwater TV surveys for *Nephrops*, provide estimates of absolute abundance derived from observed burrow densities combined with knowledge of the spatial distribution of the stock (ICES WGNEPS, 2014).

Truncated age and size structure was found to increase variability in abundance of exploited stocks (Hsieh *et al.*, 2006). This is mainly due to increased instability in the population dynamics, but is also due to higher susceptibility to environmental variability (Anderson *et al.*, 2008).

Lotka's intrinsic population growth rate (r) can be derived from an abundance time series and has been used as an indicator (Quinn and Szarzi, 1993; Jennings *et al.*, 1999; Kot, 2001; Rochet and Trenkel, 2003; Mueter and Megrey, 2005). If a linear model is fitted to the time series, then the value of the gradient (r) depends on the time period over which the model is fitted. Alternatively, a smoother (for example moving average or spline function) can be fitted to the data and the slope calculated at a particular time. A decrease in abundance, represented by negative r, could be interpreted as an effect of fishing. However, a period of reduced recruitment, potentially unconnected to the level of fishing mortality, would also result in a decline in abundance. The interpretation of trends in abundance is therefore enhanced by information on fishing effort, recruitment, length composition, or environmental conditions (Heessen and Daan, 1996). A value of r=0, implying a stable population, may be desirable as a reference point depending on whether the population is at a low or high level of abundance. At low abundance, a high positive value of r may be desirable in order to rebuild the stock. Trenkel and Rochet (2003) tested the population growth rate r as an indicator of fishing effects on Celtic Sea demersal fish populations and found the estimator to have a relatively high coefficient of variation resulting in low precision of the estimate and therefore precluding firm conclusions. Estimators of length-based indicators performed better, were more precisely estimated and had high statistical power (Trenkel and Rochet, 2003).

2.2 Catch, Landings and CPUE from Commercial Fisheries

In many data-limited situations, the only available information on which to base an assessment of stock status is commercial catch data. In the absence of other information, the interpretation of trends in catches is generally difficult. High catches may be unsustainable or sustained only by a period of high recruitment, whilst low catches could reflect either under- or overexploitation. Therefore, trends in catch data need to be interpreted in conjunction with other information, such as fishing effort data. Vasconcellos and Cochrane (2005) describe the use of catch data together with prior knowledge on population dynamics and fisheries of similar stock/species can help to infer exploitation status in data-poor situations.

The Stock Reduction Analysis (SRA) helps to reconstruct trends in stock biomass from catches by determining the past recruitment R_0 necessary to produce the current catches under the assumption of a particular stock-recruitment relationship (Kimura and Tagart, 1982; Kimura *et al.*, 1984; Walters *et al.*, 2006). In general, catches should not be used as an indicator for abundance. In cases where only landings rather than catch data are available, the interpretation of trends becomes even more difficult as temporal changes may be confounded by changes in fishing practices and/or discard rates.

CPUE derived from commercial fisheries data has been used as an indicator of abundance. However, fishery-dependent CPUE data are often highly variable and may not accurately reflect abundance when catchability changes over time. Catchability is affected by numerous factors including vessel movements, gear selectivity, season, stock structure, natural mortality, and fisheries management measures (Hilborn and Walters, 1992; Gillis and Peterman, 1998; Harley *et al.*, 2001; Maunder *et al.*, 2006). For some fleets there is uncertainty about fishing effort. Commercial CPUE data is not used for the assessment of most economically important stocks. Methods and models to standardise catch rates and catches were summarised by Maunder and Punt (2004). The year effect can be extracted by applying models that account for other factors such as area, season and vessel characteristics, and that include random variables and smoothing functions

(Generalised Linear Models (GLM), Generalised Linear Mixed models (GLMM), Generalised Additive Models (GAM)). Standardisation of fishery-dependent CPUE data by removing the vessel effect was found to improve the abundance estimates for various stocks of tuna and billfish (Carruthers *et al.*, 2011), as well as scallops (Murray *et al.*, 2013). However, commercial CPUE-based indicators should be applied with caution in particular when the relationship between CPUE and abundance is unknown (Dunn *et al.*, 2000). The interpretation of CPUE-based indicators in conjunction with size-based indicators can be more informative.

3 Spatial Indicators

Standardised fishery resource surveys can provide temporal and spatial monitoring. Although the main aim of a survey is usually to provide an index (often age-structured) of abundance for formal analytic stock assessment, survey data can also be used to reveal changes in the spatial distribution of a stock within the spatial and temporal boundaries of the survey. A variety of different indicators are available to describe the spatial distribution of stocks; for example centre of gravity, inertia, anisotropy, positive area, spreading area, and occupancy (Woillez *et al.*, 2007; Woillez *et al.*, 2009). These indicators summarise characteristics of spatial distribution, such as location, and the way space is occupied. The centre of gravity describes the mean geographic location of a stock and can be calculated for the entire stock or for subgroups (age, sex, adults). Inertia, a measure of dispersion, is calculated as the mean squared distance between individuals and the centre of gravity of the stock. Anisotropy summarises the minimal and maximal dispersion along different spatial axes. The positive area index summarises the area with catches higher than zero. The spreading area is calculated taking into account also the varying density in the area of non-zero catches. Changes in the indicators can be caused by recruitment variability, overexploitation or habitat shifts. A reduction in abundance can lead to a reduction of occupied space by the stock.

The interpretation of spatial indicators depends on the data source, i.e. survey or commercial fisheries data, and the coverage of the stock distribution area. Trenkel *et al.* (2013) suggested that occupancy (O_x), estimated as the proportion of area where x-percent of total commercial CPUE is found (for example $O_{75\%}$), should be treated with caution as an indicator of stock biomass or stock range. Spatial changes in fishing behaviour or errors in CPUE data may affect the estimate. On the other hand, Babcock *et al.* (2005) suggested that CPUE data of commercial fisheries help to relate fishermen’s behaviour to stock dynamics and support fisheries management. Other spatial indicators calculated from fishery-dependent data are the total catch per exploited area (CPEA), the exploited fraction of the ecosystem area (EFE), the mean distance from the coast of catches (MDCC), and the presence of good fishing grounds near important ports (Babcock *et al.*, 2005; Fréon *et al.*, 2005). Calculations of these indicators are facilitated using GIS methods. Changes in indicators of stock spatial distribution can be related to other biological stock indicators and stock dynamics. A local decrease or increase in stock abundance can lead to a spatial shift in exploitation of a stock. If particular stock components are reduced due to recruitment failure, overexploitation, or habitat shifts the CPUE may decrease locally or fisheries may reallocate their fishing effort to other areas. The new, more productive fishing grounds may be found further away from the ports.

4 Size-based Indicators

Commercial fishing affects the size distribution of stocks both directly and indirectly. Fisheries are size-selective and often target larger individuals reducing their numbers disproportionately. Older (and larger) individuals become fewer as cohorts accumulate the effects of fishing and natural mortality through time. Indirect effects of fishing include reduced intraspecific competition potentially leading to faster growth, earlier maturation, and increased condition of the remaining fish. Many life history processes such as reproductive output, mobility and mortality depend on size (length or weight). Indicators based on size can therefore contribute information on stock status (Blanchard *et al.*, 2005). Together with changes in CPUE, trends in size-based indicators have been related to fisheries impacts as shown for mean size of demersal fish stocks in the North Atlantic (Haedrich and Barnes, 1997).

Survey data are more likely to provide reliable estimates of size distributions, as sampling is standardised and no discarding occurs; but the estimates may be biased towards smaller individuals. Size distributions derived from commercial landings may also give useful information, in particular when fisheries target large individuals. A lack of large individuals may indicate an undesired state of the stock. Size-based indicators are easily understood, cost-effective, sensitive to fisheries impact, but are not specific to fisheries impacts alone as they also respond to other factors such changes such an environmental conditions (Shin *et al.*, 2005; Béguer *et al.*, 2012). Consideration of a combination of several size-based indicators improves interpretation, conclusions on fishing mortality, stock status and advice (Shin *et al.*, 2005).

4.1 Mean Length and Length Percentiles

Length-based indicators have been developed to determine whether stock biomass is at or above a reference point (Punt *et al.*, 2001a; Cope and Punt, 2009). Exploring a time series of length frequency data from surveys can elucidate trends in mortality, and summary statistics such as mean length or percentiles of the length distribution are also used as indicators for stock status (Trenkel *et al.*, 2007). Through sampling of commercial catches or landings, fisheries-dependent data on length frequency distributions are often available for data-limited stocks. These data can be used to assess fisheries selectivity. Since commercial catches are a subset of individuals of a stock, some information on fishing mortality and stock status may also be derived.

The mean length $\bar{L}$ and median length L_{median} of all individuals are simple length frequency summary statistics, which typically decrease at high exploitation rate (Maunder and Deriso, 2007). However, high recruitment can also reduce mean or median length initially, while the value increases again as the respective cohort ages and grows in the following years. Highly variable recruitment will result in a noisy time series, which may be difficult to interpret in terms of effects of fishing.

Beverton and Holt's average length method can be used to calculate the mean length in the catch that would result from fishing at the level of natural mortality (F=M) and is based on the von Bertalanffy growth parameters K and L_∞, length at first capture L_c (length at which 50% of the individuals are vulnerable to fisheries) and natural mortality M. $L_{F=M}$ (estimated as $0.75L_c+0.25L_\infty$) is the expected mean length in the catch when fishing mortality is the same as natural mortality and is recognised as a proxy for the mean length at MSY (ICES WKLIFE, 2012b; ICES WKLIFE, 2015). $L_{F=M}$ can be used as a reference point for the central metrics $\bar{L}$ and L_{median} in the catch, which should be at or above $L_{F=M}$ for sustainable exploitation.

The upper and lower quartiles of the length frequency distribution are likely to respond differently in relation to variation in recruitment and fishing mortality. For example, $L_{75\%}$ (the upper quartile of the length frequency distribution) may decrease with increasing fishing mortality, but remains relatively unaffected by changes in recruitment, whilst the $L_{25\%}$ (the lower quartile) can be unaffected by fishing mortality changes but would decline with high recruitment. The 95th or 90th percentile, $L_{95\%}$ or $L_{90\%}$, can be used to evaluate whether there is a truncation of the length structure (Shin *et al.*, 2005; Rochet *et al.*, 2010). A lack, in particular of large females, can indicate overexploitation and may negatively affect spawning potential. In contrast, the maximum length, L_{max}, is likely to vary with time due to the relatively low number of very large individuals, and is therefore not appropriate as an indicator for the status of larger individuals in the stock. Alternatively, the indicators mean length of the largest 5%, $L_{max5\%}$, or mean length of the largest ten individuals, L_{max10}, are less affected by stochasticity, and being based only on the right side of the length distribution are less affected by recruitment variability (Probst *et al.*, 2013a; Probst *et al.*, 2013b).

Trenkel *et al.* (2007) explored the use of length quartiles in a multiple indicator-based approach to the assessment of anglerfish species. There appeared to be neither a trend in the $L_{25\%}$ indicator (suggesting no trend in recruitment) nor in $\bar{L}$, while abundance increased. On closer inspection of length frequency distributions aggregated over many years, the peak of recruiting individuals and their respective lengths could be identified. The increase in abundance could thus be explained by a systematic increase in the number of individuals of recruiting length. This suggests that the choice of appropriate percentile summary statistics is case-specific and also that other indicators should be evaluated in conjunction. In this case, the percentage of individuals below recruiting length was used as an additional indicator for recruitment.

L_{mat} and $A_{mat,}$ the length and age at which 50% of the individuals are mature, reflect reproductive abilities and nutritional status, and are commonly used in the estimation of spawning stock biomass (Cotter *et al.*, 2009). L_{mat} is a commonly proposed reference point for the mean length in the commercial catch. At this value of mean length, theory predicts that on average enough individuals will have matured and spawned before being caught to allow sustainable exploitation (Die and Caddy, 1997). The length at first capture in commercial catches L_c indicates the length at which 50% of the individuals in a stock become vulnerable to fisheries (ICES WKLIFE, 2012b). Individuals should be allowed to reproduce at least once before capture to avoid overfishing, hence length at first capture is ideally above length at maturity (Myers and Mertz, 1998). Increased fishing mortality reducing the number of large and mature individuals may lead to reduction in mean length. A change in mean length at maturity can be caused by a purely demographic effect, removing mature individuals or cohort variability, or by a phenotypic response to stock density or environmental effects, which causes changes in the growth and maturation schedule (Helser and Almeida, 1997; Zheng, 2008).

Froese (2004) suggested three indicators related to maturity and large fecund spawners based on length composition in catches in terms of the percentage of: 1) individuals larger than L_{mat} (P_{mat}), 2) individuals at ±10% of length L_{opt} (P_{opt}), and 3) megaspawners of lengths larger L_{opt}+10% (P_{mega}). Cope and Punt (2009) further developed the concept of Froese (2004) by using standard population dynamics and life history theory to develop generic reference points for P_{mat}, P_{opt} and P_{mega}, which can be used to infer the stock status in terms of spawning stock biomass (SSB) in relation to its target reference point. Alternatively proportional stock density (PSD) or relative stock density (RSD) can be calculated as the percentage of individuals in designated length groups to describe length frequency distributions (Willis *et al.*, 1993; Neumann *et al.*, 2012):

$$\text{PSD} = \frac{\text{number} \geq L_{spec}}{\text{number} \geq L_{min}} \times 100$$

where, the minimum length, $L_{min,}$ is the lower boundary and the specified length, L_{spec}, describes the boundary for length categories of interest. L_{spec} can be defined as some percentage of maximum observed L_{max} in a catch time series, for example.

4.2 Mortality

The relationship between total instantaneous mortality rate (Z) and the mean length in the catch can be written analytically using the Beverton-Holt Z estimator:

$$Z = \frac{K(L_{\infty} - \bar{L})}{\bar{L} - L_c},$$

where K and L_{∞} are the von Bertalanffy growth parameters, L_c the length at first capture, and $\bar{L}$ is the mean length of individuals in the catch fully selected by the fishery (Beverton and Holt, 1956; Ehrhardt and Ault, 1992). It is assumed that once fully selected by the fishery, Z, the sum of natural mortality M and fishing mortality F, is constant across all lengths. Mortality, recruitment and the growth parameters are assumed to be constant over time. The mortality estimator is unbiased at equilibrium conditions, but a trend in recruitment can result in biased estimates of Z, where decreasing recruitment leads to underestimates of Z (and vice versa). Ault *et al.* (2005) investigated the bias of Beverton-Holt mortality estimates for coral-reef fishes and concluded that biases were likely to be <15% when stocks were fished at around F_{MSY}. At higher rates of fishing mortality but with similar trends in recruitment the bias would be even smaller.

A more complex relationship to relate mean length to total mortality under non-equilibrium conditions, where there have been a series of changes in the level of total mortality, has been derived by Gedamke and Hoenig (2006). They developed a transitional length statistic taking into account the time to reach equilibrium after a mortality change. The expression for mean length is fitted repeatedly to the time series of mean length data, with each of the fits assuming a different year in which a mortality change could have occurred. The transition year can be identified by maximum likelihood techniques and the respective Z levels estimated. These estimates are affected by misspecification of growth parameters, in particular of L_{∞}. The nonlinear nature of the relationship implies that changes in mortality (fishing mortality) are likely to be more difficult to detect at large mean length (closer to L_{∞}). Other estimation methods for Z, which make use of length frequency data, include the length converted catch curve (LCC), the length cohort analysis (LCA) and the Jones and van Zalinge method (JvZ), which make use of the von Bertalanffy growth parameters (Cadima, 2003). The LCC method first transforms length frequency data to age frequency data. Then Z can be estimated from a regression of catch for an age group and age corresponding to mean length (Sparre and Venema, 1998). The JvZ method directly applies a regression between aggregated catch and lengths. In contrast, the LCA estimates mortality per length class in a backward procedure which requires an estimate of mortality for the largest size class although results are insensitive to this choice. After estimating total mortality Z, changes in fishing mortality F can then be detected by including available information or assumptions on natural mortality M. The exploitation ratio is here defined as F/Z describing the fractions of deaths caused by fishing. The exploitation ratio has been suggested as an indicator for fishing pressure on the stock with a target reference point at 0.5 for fishing at MSY (Rochet and Trenkel, 2003). Both F and F/Z can be used as an indicator for the exploitation status of the stock (Radhakrishnan *et al.*, 2005; Coll *et al.*, 2006; Osio *et al.*, 2015).

Other fishing pressure indicators may be useful proxies for fishing mortality. These are usually derived from data on fishing activity such as fishing effort (days at sea), frequency of trawl events per seabed area, and fishing unit capacity (Brodziak and Link, 2002; Piet *et al.*, 2007). Such indicators should be calculated for appropriate groups of vessels, such as metiers, that have consistent fishing patterns with regard to time, space and gear. Tidd (2013) pointed out that adjusting nominal fishing effort data using, for example, generalised linear mixed models which include season and area effects, as done for CPUE or LPUE standardisation, may contribute to a better understanding of changes in fishing mortality.

4.3 Condition Indices

The condition is used to describe the nutritional status or the amount of energy reserves of an individual by relating its weight at a particular length to body composition under the assumption of allometric growth (Froese, 2006; Pinheiro and Fiscarelli, 2009). Condition is often used as an indicator of fat content, health or gonad development of individuals (Blackwell *et al.*, 2000). Condition varies seasonally in connection to spawning and food availability, and depends on individual growth rates, sex and maturity status. Changes in growth and condition can be investigated using length-weight relationships following equation:

$$W = aL^b$$

The allometric growth constant b has often been approximated by the value 3, known as the ‘cube law’ of volume increase in similarly shaped, isometric objects. In reality, this parameter differs slightly from 3 for many stocks.

The allometric growth constant can be estimated from empirical length-weight relationships and compared between areas and years or to empirical relationships, as estimated across stock range and years. The constant equals 3 when small and large individuals have comparable condition factors relative to their length. If either large or small individuals are in a worse condition than to be expected, it is assumed that values are less or greater than 3, respectively (Froese, 2006). Since many species are sexually dimorphic, the estimation should be done for each sex separately.

Instead of comparing the parameters of expected and observed length-weight relationships, Fulton's K condition index can be calculated and compared between length groups (Anderson and Neumann, 1996):

$$K = \frac{W}{L^3} \times 100$$

Alternatively, condition parameter a of the estimated length-weight relationship can be used as an indicator of condition:

$$a = \frac{W}{L^b}$$

Araújo *et al.* (2012) showed that using parameter a rather than b helped to detect decreased condition of mangrove crab *Ucides cordatus* in habitats with higher anthropogenic impacts.

Similarly, another index for condition is the relative weight of individuals, defined as the ratio of the actual weight of an individual to the weight expected from the empirical relationship for the stock (Wege and Anderson, 1978; Neumann *et al.*, 2012):

$$W_r = \frac{W}{aL^b}$$

A value of $W_r > 1$ indicates an individual is in above average condition regardless of its length. W_r can change with length as individuals undergo size-related changes such as habitat switch or maturation. The W_r-length relationship can be investigated to identify changes in environmental conditions with respect to length (Liao *et al.*, 1995). After investigation of length related changes, stock means can be calculated and compared in time or space. Blackwell *et al.* (2000) suggested using the condition (W_r) to evaluate management actions, for example to detect the effects of intraspecific competition on condition following a change in harvesting size limits.

Stocks of Atlantic cod *Gadus morhua* with higher conditions (Fulton's K) were found to have higher growth rate and recruitment potential and were thereby less vulnerable to overexploitation (Rätz and Lloret, 2003). Decreasing maturation length due to high fishing mortality was found to increase the condition (Fulton's K) of small

Peruvian hake *Merluccius gayi peruanus* (Ballón *et al.*, 2008). Immature individuals exhibited higher condition than mature ones of the same length because energy is lost with reproductive investment. On the other hand, a change in sex ratio in favour of female hake possibly caused a reduction in reproductive investment in large females and higher condition. Condition should therefore, be investigated on a stock as well as on an individual level. High condition is often linked to favourable environmental conditions in terms of habitat and food availability. *Nephrops* off the Swedish Coast were found to have higher condition in areas where they are fished by creel rather than trawl, possibly related to environmental differences between areas and differences in the impact of the gear on the habitat and the stock (Eriksson, 2006). In this study condition was calculated as the dry to wet weight ratio and the hemocyanin concentration in the haemolymph, which is the fluid circulating in the crustacean body cavity.

5 Reproductive and Morphological Characteristics

Reproduction determines the productivity of a stock, and information on changes in reproductive characteristics (age/length at maturity, sex ratio, fecundity, % mature individuals) can inform and improve fisheries management (Morgan, 2008).

Estimates of spawning stock biomass (SSB) are commonly used as an indicator for finfish. However, SSB estimates are often not available for data-limited shellfish stocks due to a lack of a quantitative stock assessment. Alternatively, a decrease in age at maturity A_{mat} was predicted and observed with increasing fishing mortality (Longhurst, 1998; Rochet, 2000), such that A_{mat} could serve as an indicator of population state (Trippel, 1995). Also a decrease in L_{mat} has been related to either high fishing mortality or to increased population density (Lizaso *et al.*, 2000; Zheng, 2008). While L_{mat} is often used to calculate reference points for central metrics of a length frequency distribution or L_c, L_{mat} can change in response to fishing mortality and may be therefore, be used as an indicator (Lappalainen *et al.*, 2016). If the minimum landing size (MLS) is determined by L_{mat}, a reduction of L_{mat} and MLS would require a concomitant reduction in the fishing mortality to protect the spawning stock (Zheng and Pengilly, 2011). The proportion of individuals above L_{mat} can be used as an indicator or alternatively the ratio of immature and mature individuals in the catch (Caddy, 2004). The proportion of mature individuals in the catch tends to decline with increasing harvest rate or in years of strong recruitment.

L_{mat} is inferred from maturity ogives which require data on maturity status at length. Maturity status of female crustaceans can be confirmed by the presence of eggs carried externally on the abdomen (berried females). The number and lengths of berried females in samples are routinely recorded. However, since a mature female is not necessarily berried, soft-shelled individuals are not sampled and berried females may be less likely to be landed, estimates of L_{mat} from landings could be biased. Alternatively, allometric relationships, including the ratio of abdominal width to carapace length have been calculated to determine maturity status of females (Lizárraga-Cubedo *et al.*, 2003). For commercial sampling of male crustaceans, other non-invasive methods are needed to determine maturity status. Changes in the relationships of the largest chela length and the carapace length can be identified, relating to differences in juvenile and adult growth (Somerton, 1980; Comeau and Conan, 1992; Goshima *et al.*, 2000; Lizárraga-Cubedo *et al.*, 2003; Hall *et al.*, 2006). However, maturity determination based on morphometric measurements may be unreliable for some species e.g. American lobster *H. americanus* (Conan *et al.*, 2001). Subtle differences in L_{mat} may only be detected at sufficiently large sample size. The cost and likely benefits of collecting additional morphometric (chelaped, abdomen) data should be evaluated for the respective stocks.

The sex ratio can be used as an indicator for fishing pressure whenever fishing is selective for one sex, for example in sexually dimorphic or sequentially hermaphroditic species (Fenberg and Roy, 2008; Fenberg and Roy, 2012). Sexual dimorphism in growth and movement may lead to selective harvesting of one sex over the other, biasing the sex ratio and differentially affecting the length structure of sexes in stocks as observed for blue crab (Carver *et al.*, 2005). In species which exhibit sexual dimorphic behaviour, catches in the fishery are often dominated by a single sex. For example, *Nephrops* trawl catches are typically dominated by males as females remain in their burrows while carrying eggs. In addition, the protection of females (or berried females) through fisheries management measures may further skew sex ratios. In a heavily exploited *Nephrops* stock at Porcupine Bank in the Celtic Sea, a change in the sex ratio (from male dominated to female dominated) in survey catches and fishery landings was observed between 2007 and 2009 (ICES WGCSE, 2013). This increased availability of females was thought to be due to generally low recruitment in previous years and high male fishing mortality (ICES, 2010). It was suggested that the reduced abundance of males led to a higher percentage of unmated females which are more likely to emerge from burrows to feed earlier in the year (like males) and thereby are available to fisheries (Stokes and Lordan, 2011).

The proportionate occurrence of individuals with certain morphological characteristics in the population has also been proposed as an indicator of adult abundance, such as the number of moulted ('soft-shell') individuals in crustacean species and shell morphology indicating maturity in certain gastropod species (Caddy, 2004). Moulting generally occurs less frequently in older animals (Caddy, 2003). Caddy *et al.* (2005) proposed that a higher proportion of animals in soft-shelled condition (after moulting) could be used as an indicator associated with increased fishing mortality, i.e. lower numbers of older/larger individuals for which moulting is less frequent. However, it should be highlighted that this indicator is sensitive to variable recruitment and, if derived from fishery-dependent data, to changes in the seasonal pattern of the fishery and discarding practices. For lobsters and crabs, a soft exoskeleton is a prerequisite for mating which occurs during a limited period after moulting when shells are soft. Therefore, a skewed sex ratio in either direction may reduce chances of finding a suitable mate and limit the reproductive potential. A skewed sex ratio and reduced size of males were found to reduce reproductive potential in crabs *Hapalogaster dentata* and spiny king crab *Paralithodes brevipes* (Sato *et al.*, 2005; Sato and Goshima, 2006). Extensive monitoring of shell lip thickness was carried out for stocks of queen conch, *Strombus gigas* in the Caribbean Sea (Appeldoorn, 1988; Ehrhardt and Valle-Esquivel, 2008). In queen conch, the growth of the flared lip is induced at the onset of maturation and data on lip thickness can provide information on changes in the mature component of the stock.

Prince *et al.* (2008) presented a novel decision tree framework which uses a rapid visual assessment of abalone stocks developed by the Victorian Western Zone Abalone Diver's Association together with trends in effort and catches to assess stock productivity and to manage the resource at the scale of individual abalone reefs. Predictable changes in shell morphology and appearance allow for determination of maturity status to assess population fecundity. Shape at onset of maturity (clean, flat shells), shape at full maturity (fouled, rounded shells) and the overall shape distribution were determined. Rounded and fouled shells are assumed to have reached adult fecundity and at least 50% of the cumulative spawning potential. As a reference point, abalone reef populations should reach at least 50% of their spawning potential to remain productive and sustain fishing, which is represented by a population comprising of a majority of rounded and fouled shells.

6 Indicators Based on Fisher's Knowledge

Questionnaires using multiple choice questions that provide information on the perceived trends in abundance, catches, CPUE and spatial stock distribution can be developed. For demersal fish stocks in the North Sea and the English Channel, fisher's knowledge was found to corroborate estimated trends in abundance, catches and catch rates from fisheries and fisheries-independent surveys (Rochet *et al.*, 2008; Macdonald *et al.*, 2014). Fisher's knowledge could be used together with fisheries data to support an inclusive approach to resource management for data-limited stocks and stakeholder participation.

The North Sea stock survey has used questionnaires to collect perceptions of fishermen from Belgium, Denmark, Netherlands, England and Scotland on changes in their economic circumstances and in the state of key stocks (Napier, 2014). This includes perception of abundance, size distribution, level of discards, recruitment, costs and profits of vessel operation. The survey results were available to ICES Advice groups. It allows for the comparison to scientific survey results, while not necessarily filling knowledge gaps in scientific stock assessments. The survey currently serves to highlight difference in some aspects in the perception by scientists, managers and fishermen's and assist a shared understanding (ICES, 2006).

7 Overview of Indicators

Table 1 summarises indicators that have been suggested or applied to fish and or shellfish stocks based on fisheries-dependent and independent data sources. Aspects covered by these indicators include spatial distribution of fish stocks and fisheries, estimates of abundance and mortality, condition and reproduction, and descriptions of length frequency distributions. The expected direction of change of each indicator in response to overexploitation is described. Indicators describing condition may respond differently to overexploitation depending on the size and sex of individuals. A careful evaluation is advised taking into account density-dependent and independent effects. Some indicators, e.g. indicators of abundance and age or length structure, can provide valuable information with regard to fishing mortality and stock status directly. Others, including those related to recruitment, fleet composition and capacity, can provide only limited information on the exploitation status when evaluated in isolation. However, indicators in this latter category can still be used as a cross-check to better understand changes in other indicators, and to exclude other factors as potential causes of undesirable changes in indicators. In general, it is advantageous to consider a range of different indicators to account for the complexity of the systems.

Table 1. List of Possible Indicators from Fisheries-dependent and -independent Data.

Data	**Indicator**	**Direction of Potential Effect from Overfishing**
Commercial fisheries (section 2.2)	landings, catch CPUE	decrease decrease
	species composition gear composition (impact)	(cross-check for changes)
	fleet days per season number traps per year and area	(cross-check for changes)
	number of vessels capacity utilization (days at sea per vessel)	(cross-check for changes)
Spatial distribution of fisheries (section 3)	catch per exploited area (CPEA)	decrease
	productive fishing grounds near important ports	decrease

	mean distance from the coast of catches (MDCC) occupancy O_x, area with x% of CPUE ($O_{75\%}$) exploited fraction of the ecosystem area (EFEA) spatial effort allocation	increase (cross-check for changes) (cross-check for changes) (cross-check for changes)
Spatial distribution of stock (survey) (section 3)	centre of gravity inertia anisotropy occupancy	(cross-check for changes)
Abundance (survey) (section 2.1)	total abundance (numbers or biomass) population growth rate (r) abundance variability	decrease decrease increase
Mortality (section 4.2)	total mortality (Z) exploitation ratio (F/Z) fishing mortality (F)	increase increase (>0.5) increase
Length frequency distribution (survey, commercial sampling) (section 4.1)	proportional length distribution (PSD) L_{max} length distribution percentile (e.g. L_{median}, $L_{75\%}$, $L_{90\%}$, $L_{95\%}$) $\bar{L}$ mean length of the largest individuals ($L_{max5\%}$, L_{max10})	decrease decrease decrease decrease decrease

	L_c (length at first capture) length proportions (P_{mat}, P_{opt}, P_{mega}) $P_{obj}= P_{mat}+P_{opt}+P_{mega}$ $P_{opt}+P_{mega}$ indicator ratios with reference points L_{mat}/L_c, $L_{opt}/L_{F=M}$, $L_\infty/L_{95\%}$ $L_{25\%}$, percentage below recruiting length (P_{rec})	decrease decrease decrease (<1) decrease increase (cross-check for changes)
Condition (section 4.3)	length-weight relationship ($W = aL^b$) indicators for individuals (K, W_r)	(cross-check for changes) change (depending on size, sex)
Reproduction (section 5)	sex ratio (males/females) % mature A_{mat}, L_{mat} % moulted (soft-shelled) individuals	(cross-check for changes) decrease decrease increase
Questionnaires to fishers (section 6)	perceived trend abundance, catches, CPUE spatial distribution	decrease (cross-check for changes)

8 Selection of Indicators

The selection of indicators is usually a step-wise process (Rice and Rochet, 2005). To begin with, needs of users and objectives have to be identified, and a list of candidate indicators compiled. Proposed indicators should be validated before forming the basis of management advice. A number of authors (Halliday *et al.*, 2001; Rice, 2003; Rochet and Trenkel, 2003; Rice and Rochet, 2005; Rees *et al.*, 2008; Cotter *et al.*, 2009) have advocated the completion of a template to summarise and compare the properties of potential indicators and help with the process of indicator selection as follows:

- **Indicator name**
- **Characteristic -** the characteristic the indicator reflects, e.g. abundance, production, or an ecosystem property.
- **Concreteness -** direct observation or model output; biological property or abstract concept.
- **Theoretical basis -** acceptance, support, evidence, scientific defensibility.
- **Reference Points -** to categorise stock (or ecosystem) states on the basis of the indicator value.
- **Estimation & measurability -** a description of how the indicator is calculated including data sources, range selection, transformations and time series smoothing.
- **Accuracy and precision -** how well does the indicator reflect the actual state? What are the statistical properties (variance in measurements, bias, direction of bias etc.)?
- **Interpretability -** description of how the indicator reflects the identified stock status characteristic.
- **Availability of historical data**
- **Cost -** data already available, costly instrumentation or data collection
- **Representability -** can the indicator be used to generalised beyond the period or location analysed (season, larger geographic area)?
- **Sensitivity -** is it possible to identify small changes in the indicator value. Is natural variability likely to mask change in the stock characteristic?
- **Responsiveness -** time scale at which changes in management are likely to be mirrored in the outcome in terms of the indicator value.
- **Specificity -** is it possible to disentangle effects of fishing from other impacts.

- **Robustness -** is the meaning of the indicator value robust to errors in underlying assumptions?
- **Easy to communicate -** depends on target audience, e.g. general public, managers, fisheries.

The relative importance of the different selection criteria as well as the perception of relevance and utility of an indicator will depend on the people involved in the selection process and on the purpose of the indicator. In many cases, however, the suggested template is likely to be incomplete due to a lack of information on exactly how the indicator responds and is related to changes in stock status. Once objectives have been identified, indicators are selected and reference values are set accordingly, methods to aggregate, visualize and evaluate individual or multiple indicators can be applied.

9 Evaluation of Indicators

9.1 Single Indicators

Conclusions on indicator change can be drawn via hypothesis testing comparing the indicator value to a reference point. However, it can be useful instead to detect trends in indicator time series to decide whether fishing will decrease or increase the value of an indicator (Jennings and Dulvy, 2005; Rochet *et al.*, 2005). Trend-based approaches rely on knowing (i) the status of the stock at the beginning of the time period, and (ii) whether it has improved or deteriorated since then. The results will depend on the chosen time window, as trends in recent years may differ from long-term trends. The estimation of linear trends facilitates an extrapolation to future years but ignores any non-linear fluctuations in the indicator. Therefore, alternative methods including fitting non-linear smoothers through data series may be useful (Nicholson and Fryer, 2002; Holmes *et al.*, 2008; Blanchard *et al.*, 2010; Large *et al.*, 2013). Given that empirical indicators are derived from sampled data, in situations with poor or erratic sampling levels the signals in these indicators may be difficult to distinguish from noise in the data. The provision of qualitative indicator-based advice relies on the correct identification of these signals.

Other data-limited approaches relate to quality control methods, such as Shewart, moving average and CUSUM control charts (Scandol, 2003; Scandol, 2005). In these methods, indicators are flagged in case of uncharacteristic values or trends. Shewart control charts are only useful to detect short term changes, because there is no memory of past events. To detect persistent changes, a moving average control chart or a cumulative sum (CUSUM) can be used. The moving average control chart calculates means of the last x observed indicator values, and observations are compared against a decision interval of the smoothed series. With help of the cumulative sum (CUSUM) control chart positive and negative deviations in an indicator from a reference mean are summed over time and flagged when the sum leaves a decision interval (Scandol, 2003; Mesnil and Petitgas, 2009). CUSUM is effective in the detection of changes with strong variability in the data. The selection of an appropriate reference value is critical (ICES WGMG, 2008). For cases where no long-term reference mean is available, the Self-Starting CUSUM (SS-CUSUM) was developed. This method uses the running mean based on observed indicator values and is updated as new data become available, excluding possible outliers (Pazhayamadom *et al.*, 2013). CUSUM can be used to formulate harvest control rules (Pazhayamadom *et al.*, 2015).

Statistical power analysis relates to the type II error of hypothesis testing, the probability of accepting H_0 given it is false. Thereby, the probability of detecting a particular magnitude of change in a sampling setup can be quantified (Peterman, 1990). Alternatively, power analysis can be used to determine the magnitude of change or trend which could be detected in a defined number of years at a particular level of confidence and targets can be set accordingly (Peterman, 1990; Nicholson and Fryer, 2002). The analysis can also be formulated to quantify the sample size (or number of years) necessary to detect a defined change in a variable with a defined level of confidence (Nicholson *et al.*, 1997). If the power is low, then it may take many years of data collection before a trend can be identified or may require the modification of the data collection process or lead to the choice of another indicator. Nicholson and Jennings (2004) analysed size-based indicators of the North Sea fish community using the IBTS (International Bottom Trawl Survey) data and concluded that at least 10 years of data were required to detect trends in these indicators. Use of such indicators may therefore lead to managers failing to identify a necessary management action.

Another potential difficulty is the possibility of making type I errors – rejecting H_0 and concluding that there is a trend in an indicator when there actually is none. This could result in advice and subsequent management action being based on noise rather than a true signal in the data. The seriousness of potential implications for the stock and fishing industry of making errors of judgement regarding trend analysis depends on the type of error (I or II) and type of indicator being analysed. It is therefore important that these issues are taken into account when considering significance levels and acceptable power of statistical tests.

9.2 Combining Indicators

Given that changes in stocks may be reflected in various characteristics and that a single indicator may not be sufficient to detect changes, the use of multiple indicators and/or combinations of indicators is often advocated. Rice and Rochet (2005) review methods for combining indicators and suggest a three stage procedure comprising: i) standardisation, ii) weighting and iii) combining. Standardisation may consist of converting an indicator into discrete values or linearly interpolating on a common range or between reference values. The indicators can be combined either graphically using kite or radar diagrams (Garcia and Staples, 2000; Fay *et al.*, 2013; Fulton *et al.*, 2014) or numerically (weighted/non-weighted average or sum, arithmetic or geometric mean, fuzzy numbers). Petitgas and Poulard (2009) summarised multiple spatial indicators in a single multivariate indicator using a multifactor analysis (MFA); any deviation from reference means are detected using the CUSUM control chart.

ICES (2012) suggested that advice for data-limited stocks could be based on a Productivity and Susceptibility Analysis (PSA) and the landings of the previous year. PSA is a semi-quantitative assessment of the rate at which a stock can recover from depletion (productivity) and the potential of the stock to be impacted by fisheries. The PSA was initially developed for bycatch species in Australian prawn fishery and is based on scoring life history information and knowledge about fishing mortality to assess expected relative vulnerability (Stobutzki *et al.*, 2001). Multiple indicator scores relating to productivity (growth rate, L_{max}, L_{mat}, natural mortality, fecundity, reproductive strategy, trophic level) and susceptibility (area, fisheries impact on habitat, concentration, migration, behaviour, morphology, catchability, fishing mortality, SSB, discard mortality, commercial value) are weighted and combined. The vulnerability index is calculated using Euclidean distances between the scores and the theoretical scores of lowest vulnerability, i.e. high productivity and low susceptibility (Milton, 2001; Stobutzki *et al.*, 2001; Field *et al.*, 2010; Patrick *et al.*, 2010; McCully Phillips *et al.*, 2015). The index gives a preliminary indication of the risk of overfishing of a stock which helps to identify the level of precaution that should be adopted when giving management advice (Osio *et al.*, 2015).

Rochet *et al.* (2005) chose a ‘time trend approach’ to combine the analysis of trends in a pair of indicators ($\bar{L}$ and log(abundance)) in exploited fish populations in relation to an initial state. Trends in indicators were analysed separately and subsequently combined (Rochet *et al.*, 2005). This was done using two-dimensional diagnostic tables which combine the outcomes of the indicator pair, their interpretation and potential mechanisms. The interpretations of possible combinations were based on their biological significance, whether trends could have been caused by fishing and whether trends could be reversed by reduction of fishing mortality (Rochet *et al.*, 2005). The analysis of indicators was based on a historical data series and probabilities for each parameter combination were calculated. Alternatively, two correlated indicator time series can be combined to a single indicator and then jointly assessed (de la Mare and Constable, 2000; Reid *et al.*, 2005). The combined index was found to provide a better fit to the data than the individual ones (Reid *et al.*, 2005).

Brind'Amour and Lobry (2009) combined multiple indicators for the assessment of the ecological status of an estuary comparing two methods, the time-trend approach and the single Multi-Metric Index (MMI). It was shown that the two methods can lead to opposite conclusions regarding the exploitation status of a stock or a community; mainly because reference levels differed and each method addressed different ecological aspects. While the time-trend approach is based on a historical time series, comparing the indicator to a single experience-based reference value, the MMI method is based on a spatial comparison with multiple reference areas using a virtual maximum as a reference value.

The ‘traffic light approach’ (TLA) describes a management framework using colours to categorise and review the state of multiple indicators (Caddy, 1999; Halliday *et al.*, 2001; Caddy, 2002; Caddy, 2004) and has proved to be popular with the fishing industry and fishery managers as a format for the basis of discussions on stock status (Koeller *et al.*, 2000). Caddy (1999) proposed a series of over 30 quantitative and qualitative indicators whose current values could be scored against reference points to determine whether they have ‘red’, ‘yellow’, or ‘green’ status - ‘red’ being associated with concern about the current and/or future stock status, ‘yellow’ an intermediate level representing uncertainty about the future status and ‘green’ with a positive outlook. Once assigned, colour codes of the various indicators are combined to produce an overall ‘summary’ score. A simple version of the TLA is used in ICES advice to summarise stock status with regard to fishing mortality and stock biomass relative to available reference points.

Caddy *et al.* (2005) suggested that the colour boundaries of the TLA are equivalent to the respective precautionary (green-yellow boundary) and limit (yellow-red) reference point as defined by ICES. However, for many stocks the exact values of these boundaries are unknown. In their assessment of northern shrimp (*Pandalus borealis*) on the eastern Scotian Shelf, Koeller *et al.* (2002) instead chose the 66th and 33rd percentiles of the indicator values in the time series as default colour boundaries. In this approach the colour boundaries will change with the addition of new data points, which implies that the perception of historical stock status may also change. In contrast, for Gulf of St. Lawrence snow crab (*Chionoecetes opilio*) the colour boundaries were defined by dividing the indicator value range into equal sections, such that the boundaries are only affected when adding data points outside the previously observed range (Caddy *et al.*, 2005). Indicators can be grouped according to one of three characteristics: abundance, production (recruitment and growth), and fishing mortality (Koeller *et al.*, 2002). Indicator values can also be presented individually. An evaluation of status can then be made following, for example the 'one out - all out' approach, where a single indicator with a bad score leads to overall negative evaluation. Alternatively, pre-agreed management actions may become gradually more severe (for example in terms of effort limits) with increasing number of indicators pointing towards a deterioration.

Koeller *et al.* (2000) showed using simulation testing for a northern shrimp population model that it is possible to link the combined TLA scores to formal management decision rules. In the scenarios tested, they found that a harvest rule based on the combined score from a greater number of indicators performed better in terms of both yield and risk to the stock. Indicators, scoring between -1 and 1, were combined taking the arithmetic sum. The yellow category (scores close to 0) should be interpreted with caution whenever it does not necessarily represent an intermediate condition but rather uncertainty. Combined indicator scores can be calculated using various aggregation metrics (e.g. the weighted or unweighted sum, arithmetic or geometric mean or product).

The decision tree approach allows for informed management decisions based on a consistent method taking into account uncertainties (Peterman, 2004; Wilson *et al.*, 2010; Link *et al.*, 2011). Prince *et al.* (2011) suggested the use of a decision tree approach combining indicators such as size-dependent catch rates and the proportion of old fish for the Australian longline fishery. Multiple indicators are evaluated using a rule-based model with classical Boolean logic or 'fuzzy set theory' (Jarre *et al.*, 2008). The Boolean knowledge-based system sets thresholds to evaluate indicators which are then weighted to reach a decision (Miller and Field, 2002). The 'fuzzy set theory' is also based on heuristic rules which are formulated into sequences of 'if-then' clauses that lead to an overall conclusion. The 'fuzzy set' boundaries are not necessarily sharp and allow for gradual membership (Cheung *et al.*, 2005; Paterson *et al.*, 2007).

10 Indicators in Management Strategy Evaluation (MSE)

It is generally difficult to predict how well indicators reflect actual changes in the status of data-limited stocks. Stock abundance is unknown and often cannot be estimated directly. Furthermore, it is hard to deduce whether selected indicators, reference points and harvest control rules are likely to achieve management objectives.

Management Strategy Evaluation (MSE) is a modelling framework that enables testing of alternative management options, either before implementation or retrospectively, to consider whether implemented harvesting strategies should be revised (Sainsbury *et al.*, 2000; Edwards and Dankel, 2016).

MSE is a quantitative approach and takes into account a range of potential states of the world and uncertainty (Smith, 1993; Smith *et al.*, 1999; Ives *et al.*, 2013). An 'operating model', capturing the key population dynamics of a given stock, is combined with a 'monitoring model' and a 'management model', which includes HCRs, generating data with realistic measurement uncertainty (bias and variance). The resulting data are processed to calculate performance measures which are then compared among management scenarios and to 'true' values if available (Punt and Hobday, 2009). Management strategy evaluation allows for decision-making taking into account possible future catches as well as resource risks (Butterworth and Punt, 1999; Geromont *et al.*, 1999).

MSE can be used to test whether suggested harvest control rules will achieve management objectives for data-limited stocks. Harvest control rules link information on stock abundance to catch limits. It can be tested whether selected indicators can identify if a stock status is close to undesirable states (Punt *et al.*, 2001b). The MSE allows for the selection of appropriate indicators used in harvest control rules. MSE was used to test low-information, low-cost harvesting approaches with rotational harvesting cycles of varying duration for the sea cucumber fishery (Plagányi *et al.*, 2015). The MSE is suitable to test indicator-based decision rules (Sainsbury *et al.*, 2000). Population-level indicators for a variety of species in south eastern Australia were found to be more sensitive to short-term fluctuations and species-specific effects than community or ecosystem indicators (Fulton *et al.*, 2005). Whenever, environmental variability strongly affects indicator values, a multivariate analysis can be applied (Link *et al.*, 2002).

Lehuta *et al.* (2013) tested the suitability of different indicators for the management performance of a pelagic mixed fishery. For anchovies, the biomass level relative to

B_{lim}, total landings, as well as fisheries effort distributions of particular fleets were shown to be indicators sensitive to management measures (Lehuta *et al.*, 2013). Potential effects of changes in productivity for example caused by environmental variability or climate change can be included in the MSE, usually by assuming a frequency and duration of regime shifts in recruitment, changes in natural mortality, growth or carrying capacity (Kell *et al.*, 2005; Szuwalski and Punt, 2013; Punt *et al.*, 2014). Harvest control rules based on regime shifts have been formulated and tested in a MSE for the eastern Bering Sea snow crab fishery (Szuwalski and Punt, 2016). It was concluded that the regime-based HCR should be applied only when the stock actually is subject to a regime-based shift (Szuwalski and Punt, 2016). It can, however, be difficult to identify stocks subject to regime shifts and whether a shift is actually due to environmental variability (Szuwalski and Punt, 2013).

MSE has been applied to data-poor stocks with available catch-at-age data, to test harvest control rules based on a target fishing mortality (Wayte and Klaer, 2010). CPUE-based harvest control rules have been evaluated, where it was shown that the appropriate selection of a reference period (historical or current catch) is critical to the outcome (Little *et al.*, 2011).

Also, size-based indicators have been tested with the MSE framework. Mean length of fully selected fish was used to estimate fishing mortality and evaluated in a harvest control rule which adjust a reference catch for Australian temperate demersal species (Klaer *et al.*, 2012). The method performed reasonably well. Size-based indicators can also enter control rules directly. Punt *et al.* (2001a) evaluated the use of indicators in management for broadbill swordfish using Monte Carlo simulations. The size-based indicators such as the $\bar{L}$, $L_{95\%}$, and mean weight ($\bar{W}$) performed better than indicators based on catch rates and relate more predictably to stock abundance. Indicators were calculated as the average of previous years to reduce variability. A harvest control rule based on $\bar{L}$ and $L_{F=M}$ as a reference point were tested using MSE for a wide range of species and life histories (Jardim *et al.*, 2015). This control rule succeeded in reversing decline in biomass. However, the control rule was sensitive to life history characteristics of the respective stock (L_{mat}, L_{∞}) relative to fisheries selectivity.

11 Examples of the Use of Indicators in Assessment, Advice and Management of Shellfish

There are few shellfish stocks for which formal decision rules for management actions based on indicators have been agreed, and even fewer for which these rules have been formally simulation tested. Some examples of the use of indicators in shellfish fisheries management, including species relevant to Scottish fisheries, e.g. lobsters, scallops, crabs and *Nephrops,* are described below.

Table 2. Case Summaries of Indicator Use in Assessment and Management for Crustacean and Molluscs Stocks.

Stock	**Area**	**Indicator**	**Assessment**	**Simulation-tested**
Panulirus cygnus, western rock lobster	Western Australia	egg production, SSB, survey larval settlement index	yes	yes
Jasus edwarsii, southern rock lobster	- South Australia - Victoria, Australia - New Zealand	- CPUE, pre-recruit index - biomass estimate, stock rebuilding rate, CPUE - CPUE	- yes - yes - yes	- yes - yes - yes
Jasus landii, west coast rock lobster	South Africa	CPUE, somatic growth data, fisheries independent monitoring index	yes	yes
Homarus americanus, American clawed lobster	east Canada	median carapace length, CPUE, catch rate of berried females, landings	yes	no
Scylla serrata, giant mud crabs	Northern Territory, Australia	mean size (carapace width), commercial effort, total commercial catch	yes	yes

Chinoecetes opilio, snow crab	Atlantic Canada	% soft-shell crabs, effort, landings, CPUE, recruitment	yes	yes
Aequipecten opercularis, Queen scallop	- Faroe Islands - Isle of Man	- CPUE - CPUE	- no, CPUE in a "move-on-rule" - yes	- no - no
Plagopecten magellanicus, Atlantic sea scallop	Canada	abundance estimate, recruitment, capacity utilization, value of landings, catch per sea day, mean size	yes	yes
Nephrops norvegicus	Northeast Atlantic	density (TV-survey), CPUE, length frequency indicators	yes	no
European lobster, king scallops, velvet crab, brown crab	Shetland Islands, UK	LPUE, number undersized individuals, % mature large individuals, mean carapace length, sex ratio	yes	no

11.1 Lobster and Crab

11.1.1 *Panulirus cygnus,* Western Rock Lobster, Western Australia

The management framework for western rock lobster in Western Australia aims to keep egg production above a given threshold to ensure sustainability. An index for annual egg production is estimated for each management zone accounting for the effect of site and environmental conditions from fisheries independent surveys. The target reference points for egg production as well as spawning stock biomass were recommended at the levels estimated for 1980, which represented 25% of the unfished stock (Hall and Chubb, 2001). To achieve these targets, berried females were protected, pot quotas were reduced, and maximum landing sizes for females were introduced. A larvae settlement index from surveys is included as an important component of the assessment. Settlement of rock lobster larvae along the west coast of Australia was found to be highly correlated with environmental conditions relating to water temperature, current strength and spawning stock biomass vulnerable to fisheries (Caputi *et al.*, 1995b; Caputi *et al.*, 2001; Caputi, 2008). Data on larvae settlement were used to predict recruitment and catches three to four years ahead (Caputi *et al.*, 1995a). In 2008 and 2009, the observation of relatively low larvae settlement led to the implementation of effort reductions of 44% and 73%, respectively (Reid *et al.*, 2013). Reid *et al.* (2013) tested different effort scenarios and their economic impact using an assessment model to maximise economic yield and showed that these effort reductions in 2008/2009 ensured a profitable fishery in the following five years.

11.1.2 *Jasus edwardsii*, Southern Rock Lobster, Australia and New Zealand

The decline in biomass and recruitment of southern rock lobster (*J. edwardsii*) off the coast of South Australia in the 2000s, led to a reduction of TACs in several management zones. A management plan for the Northern and Southern Zone set up in 2007 used standardised CPUE of legal-sized rock lobster and the pre-recruit index (the number of undersized individuals per pot lift) as the key performance indicators (Sloan and Crosthwaite, 2007). The CPUE in 2004, the year when fishing pressure was lowest, was selected as a limit reference point. For the pre-recruit index, the mean of the period 1995-2004 was selected as a reference point. A decision control rule which reduces the TAC by at least 10% was triggered whenever both indicators dropped below their respective reference points. If both indicators were above the reference level the TAC may be increased up to 10% (Sloan and Crosthwaite, 2007). In 2010, the fishery stakeholders reviewed the harvest strategy and proposed two new modified decision rules specifically regulating the TAC following a constant exploitation rate policy to counteract the decline in the stock. Out of the two rules, a “discrete” rule (discrete with four possible outcomes for TAC), initially proposed by the fishing industry, was found to restrict extremes of TACs and was subsequently implemented with support of the stakeholders in the new management plan in 2011 (Punt *et al.*, 2012). The implemented discrete HCRs for northern and the southern zone rock lobster were based on the most recent catch rates with a defined limit reference point, a pre-recruit index threshold, and a cap on maximum TAC and showed optimal economically optimal performance (Linnane *et al.*, 2014; McGarvey *et al.*, 2014).

A number of studies of southern rock lobster *J. edwardsii* off Victoria, Australia, illustrate the use of commercial fisheries data, including individual sizes and maturity status and modelling tools for lobster assessment and management (Hobday *et al.*, 2005; Punt *et al.*, 2006a; Punt *et al.*, 2006b; Punt *et al.*, 2013b). So far, the results of the stock assessment have been used as input to the management process (DPI, 2009). Since 2011, the TAC has been set annually according to a hierarchical management decision framework, combining a stock assessment with the use of indicators (DPI, 2009). The framework includes decision rules for the TAC which utilise the output from stock assessments (biomass estimates), the projected stock rebuilding rate and the trend in observed standardised CPUE, in the order mentioned.

The rock lobster fishery has been the most commercially important fishery in New Zealand since the 1950s (Annala, 1983). Landings decreased in the late 1980s and in 1991 a quota-system was implemented (Miller and Breen, 2010). The stock assessment involves running a highly complex Bayesian model which requires substantial resources and is therefore only carried out intermittently (Bentley *et al.*, 2005). Annual management advice is based on management procedures and harvest control rules using standardised CPUE (catch/pot lifted) as an indicator. The harvest control rules and their complexity vary between sub-stocks (Holland *et al.*, 2005; Breen, 2009; NRLMG, 2014). For rock lobsters in areas CRA7 and CRA8, a management procedure was implemented in 1997 with annually set TACs, which follow decision rules based on a linear relationship of observed commercial CPUE and TAC. The harvesting strategies were successful in recovering biomass in these areas (Haist *et al.*, 2009; Miller and Breen, 2010). However, for the CRA 4 area, only when CPUE decreased substantially (in 2006 and again in 2008) and the TACs could not be fully utilised, did the stakeholders agree to a further reduction of TACs following the results of simulation-tested management procedures (Breen *et al.*, 2006; Breen *et al.*, 2009).

11.1.3 *Jasus lalandii,* West Coast Rock Lobster, South Africa

South African West Coast rock lobster *J. lalandii,* exhibited a severe decline in stock biomass and somatic growth in the late 1980s, and a management procedure was developed in 1997 (Cockcroft *et al.*, 2008). A working group developed a size-structured operating model incorporating somatic growth data in order to assess stock status and evaluate alternative CPUE-based management strategies (Johnston and Butterworth, 2005). Candidate procedures were tested to evaluate whether the biomass of individuals above 75 mm (B_{75mm}) would increase within 10 years of implementation. One of the candidate procedures, potentially leading to biomass recovery and allowing for an increase in TAC and effort, was selected and implemented in 2003. This procedure was designed to allow for a fast, but still constrained, response in TAC to changes in recruitment and somatic growth, and was aimed to achieve a relatively small increase in B_{75mm} (Johnston and Butterworth, 2005).

11.1.4 *Homarus americanus*, American Clawed Lobster, East Canada

Fisheries on American lobster, *H. americanus*, off the Canadian east coast are currently managed by landing size limits and the protection of mature females. The Canadian Atlantic lobster fisheries are controlled through limited licences, first introduced in 1967, constraining capacity when abundance increases. On the basis of indicator trends, including a 'fairly constant' median carapace length and catch rates of berried females over a period of 15 years, Miller and Duggan (2004) concluded that the lobster stock in the Canadian East Coast (Nova Scotia) could support increased catches. The lobster fishing effort in Canada was controlled and held relatively constant (Miller, 2003; Steneck and Wahle, 2013). Since the collapse of large finfish stocks in the northwest Atlantic, lobster stock biomass has generally been high which is thought to be due both to low natural mortality in the absence of large predators and the offshore expansion of the species' range leading to higher larval supply (Boudreau and Worm, 2010). In recent years, there has been some reduction in the abundance of the largest lobsters while CPUE and landings remained high (Tremblay *et al.*, 2013).

11.1.5 *Scylla serrata,* Giant Mud Crabs, Northern Territory, Australia

Giant mud crabs, *S. serrata,* inhabit mangroves and estuaries in the Indo-Pacific Ocean and are extensively harvested. Mud crabs are characterised by rapid growth, early maturity, high fecundity and short life span. In Northern Territory, Australia, a mud crab fishery management plan was introduced in 1991. Increasing catches in the 1990s were followed by dramatic declines from 2001 to 2003 (NTG, 2013). Declining mean carapace width together with declining catches and catch rate supported the advice that giant mud crab were fully exploited and that fishing effort should not be allowed to increase further (Haddon *et al.*, 2004). Following the 2004 assessment and considering the large decline in catches, the minimum landing size was increased by 10 mm in 2006. Total commercial catch, commercial effort, and the overall mean size of crabs in the catch were used as crab fisheries performance indicators in the 2007 assessment (Ward *et al.*, 2008). The Beverton-Holt mean length estimator is used to provide estimates of the long-term trend in total mortality rate from changes in mean carapace width. Seasonal dynamics are captured using a size-age-sex monthly stock synthesis model (Grubert *et al.*, 2013). While in the 2007 assessment fishing mortality appeared to be high, in the 2012 assessment, it was concluded overfishing risks were low, but that there is a decline in maximum carapace width. However, since the increase in MLS in 2006, the seasonal pattern in mean carapace width has stabilised, halting a decline which started in the early 2000s (Grubert *et al.*, 2013).

11.1.6 *Chinoecetes opilio,* Snow Crab, Atlantic Canada

Snow crab, *C. opilio,* along the Canadian Atlantic coast has been managed as a male-only fishery with a MLS to ensure pre-capture mating. Stock biomass has been fluctuating over the years, reaching a very low level in the late 1980s. In 1984, an individual vessel quota system was implemented. In 1990, temporary seasonal/spatial closures limited the fishery to protect soft-shell crabs found in large amounts in the catches (Caddy *et al.*, 2005). Several years of high recruitment led to an increase in biomass in the 1990s, followed by a decline to a generally low level until recently. In 2004, a soft shell crab protocol was introduced, such that specific areas can be closed anytime when the percentage of soft-shell crabs in at-sea samples remains above 20% for a certain number of days to protect this component of the stock with (temporarily) little commercial value and high discard mortality (Siddeek *et al.*, 2004; Mullowney *et al.*, 2014a).

An indicator approach taking into account fisheries (effort, landings, CPUE, percentage of soft-shelled crabs), abundance and recruitment data was developed by Caddy *et al.* (2005). The approach confirmed a critical period in several indicators in the late 1980s and late 1990s over a period when commercial landings were low.

Snow crab in Newfoundland and Labrador are managed using indicators such as commercial CPUE, exploitable and pre-recruit biomass estimated from surveys, the proportion of soft-shell crabs and mature females carrying eggs (Addison *et al.*, 2013). Management actions, such as closures, are triggered if the proportion of soft-shell crabs in samples taken at sea reaches the threshold of 20% (Addison *et al.*, 2013). Productivity of snow crab stocks is still low, primarily due to low recruitment caused by a warming oceanographic regime (Mullowney *et al.*, 2014a; Mullowney *et al.*, 2014b).

11.2 Scallop

11.2.1 *Aequipecten opercularis*, Queen Scallop, Faroe Islands and the Isle of Man

A small-scale fishery for queen scallop, *A. opercularis,* in the Faroe Islands started in 1970. With depletion of local scallop beds, the number of vessels decreased, and since 1988 only one vessel with three licences for separate areas has prosecuted the fishery. No further licences have been issued due to concerns over sustainability and profitability. Queen scallops from the Faroe Islands contribute 20-30% of European landings of this species. Rather than being regulated by TACs, management is based on effort limitation, restrictions on fishing days, and a 'move-on rule' when CPUE decreases (LeRoux *et al.*, 2013). Currently, the fishery is estimated to occupy about 9% of scallop grounds of the Faroe Islands, and is considered to be sustainable with relatively low but stable CPUE.

Queen scallops around the Isle of Man are caught using trawls and dredges. The stock is managed according to a biomass index derived from surveys and commercial CPUE, and using management measures such as TAC, MLS, fishing exclusion zones, and gear restrictions. Significant relationships between CPUE from logbooks and abundance estimated from fisheries-independent surveys were shown, such that trends in commercial CPUE were considered as reliable stock proxies for abundance (Andrews *et al.*, 2011). The otter trawl scallop fishery was accredited by the Marine Stewardship Council (MSC) in 2011 but the certification was subsequently suspended in 2014 due to low stock levels. In contrast, the dredge scallop fishery has a strong negative impact on benthic habitat, was suggested to be banned (Murrray *et al.*, 2009; Hinz *et al.*, 2012) and failed the MSC accreditation (Andrews *et al.*, 2011).

11.2.2 *Plagopecten magellanicus*, Atlantic Sea Scallop, Canada

Repetto (2001) described the management of Canadian Atlantic sea scallop (*P. magellanicus*) fisheries. In 1973, a mean size restriction was introduced. After a decline in abundance in the early 1980s, fishing effort was reduced, and the fishery was able to support high catches at lower fishing rate. The stock biomass started to recover in the late 1980s and is now considered to be at a healthy level (DFO, 2013). The exploitation rate has been at a stable low level since 2000. The Canadian fishing industry makes a voluntary contribution to the cost of the government's research surveys which provide data on age and size distribution on a fine spatial scale. Annual catch limits are set conservatively as a proportion of scallop abundance as estimated from the sample survey. Catch limits are adjusted to account for the level of recent recruitment, such that TACs are reduced if the survey indicates weak incoming year classes. The higher number of older large individuals in the stock attenuated the effects of recruitment variability on adult biomass (Repetto, 2001). In addition to biological indicators, such as capacity utilization (days at sea per vessel), the value of landings per tow and the catch per sea day were used in the study as economic indicators for profitability. Capacity utilization increased, catch per sea day has risen, and the value increased due to higher abundance of large scallops (Repetto, 2001).

11.3 *Nephrops norvegicus*, Northeast Atlantic

Nephrops inhabit burrows in muddy sediment and emerge to feed. Males leave their burrows more frequently than egg-bearing females, leading to male-dominated catches. *Nephrops* are caught using creels or trawls. The status of a number of *Nephrops* stocks across the Northeast Atlantic region, including Scottish *Nephrops* stocks, is assessed using underwater TV surveys. Observed density raised to the suitable sediment area provides an abundance estimate. So far, there are no precautionary reference points defined for stocks, and no formal management plans have been adopted. Quantitative advice is provided on the basis of the most recent abundance estimate and an F_{MSY} proxy derived from per-recruit analyses using fisheries data (ICES WGNSSK, 2010). Currently, landings are restricted by TACs at the level of ICES subarea, with each TAC covering a number of different *Nephrops* stocks (EC, 2013). The rules for providing advice require further simulation testing to investigate their sensitivity to assumptions on discard rates and mean weights. Indicators used for *Nephrops* stock assessment and advice include mainly a survey abundance index (if available), commercial standardised CPUE or LPUE, and those based on the length frequencies (ICES, 2013). Length frequency indicators are the proportion of large individuals, slope of the right side of the length frequency distribution, and mean size.

11.4 Shellfish Stocks in Scottish Waters

Although the Scottish shellfish fisheries (with the exception of *Nephrops*) are not regulated by TACs, regional assessments are conducted for other crustacean species including stocks of brown crab, velvet crab and European lobster (Mesquita *et al.*, 2016). For these, length frequency data from commercial landings sampling are used in a Length Cohort Analysis (LCA) to estimate fishing mortality. Length is measured as carapace length for lobsters and as carapace width for crabs. Biological parameters are estimated from both historic tagging studies and commercial sampling data. Also recently, the average size of the largest 20% of individuals has been investigated as a potential indicator of stock status. No formal decision rules or management actions are associated with the crab and lobster assessments.

In Shetland, management of fisheries for European lobster, king scallops *Pecten maximus*, velvet crab and brown crab use LPUE as an abundance indicator. Target reference points for each species are based on the running average of the period 2002-2009 (Hervás *et al.*, 2012). Various management measures are triggered by a range of limit reference points. When the LPUE indicates stock status deterioration, measures including a limit on new licences, area closures, ban on landing of berried females, and changes in minimum and maximum landing sizes can be implemented. In addition, depending on the species, various other indicators can be used to evaluate stock status and provide advice, i.e. the number of undersized individuals, proportion of mature or large individuals, length frequencies, and mean carapace length as indicators of overfishing, and the sex ratio as an indicator for reproductive potential (Hervás *et al.*, 2012). The management framework operative under the Shetland Island Regulated Fishery (Scotland) Order 2009 and the status of Shetland scallop, brown crab and velvet crab stocks and the respective fisheries were deemed adequate for the fisheries to gain MSC accreditation in 2012 (Hervás *et al.*, 2012). The MSC certification report, however, includes a comment on the appropriateness of the reference points and their lack of biological basis for the evaluation of stock status as well as a condition attached to the accreditation requiring the development and adoption of biologically-based limit and target reference points by year five.

11.5 Other Data-poor Fish and Shellfish Stocks

Sadykova *et al.* (2009) developed a population model for European crayfish *Astacus astacus*, a freshwater decapod, using CPUE data from commercial fisheries data to estimate the size of the exploitable population and sex ratio and fecundity estimated from annual test fishing. The model was used to explain population decline in the late 1980s. The model combines discrete growth and the effect of temperature on moulting frequency which could also be applied as a predictive model for management of other crustacean species.

The Tasmanian dive fishery for sea urchins (*Heliocidaris, Centrostephanus*) and periwinkles (*Turbo sp.*) is managed using CPUE and relative abundance of cohorts in commercial catch samples as indicators (Anon., 2005). Constant TACs have been set conservatively at 75% of the mean historic catch in a reference period (2000-2004). Management action is triggered by two performance indicators CPUE and catch composition; i.e. when CPUE declines by 35% in any one year or by 20% in two consecutive years or when there are undesirable changes in catch age composition. Possible management actions include closures and a change in the size limits.

The abalone dive fishery of Tasmania lands two species, mostly *Haliotis rubra* (blacklip abalone) and some *Haliotis laevigata* (greenlip abalone). Following a period of heavy exploitation in the 1970s and early 1980s, when an increase in catching efficiency caused a severe decline in stock abundance, quota reductions led to more stable catches and recovery of the stocks. CPUE, catches and the length quartiles are used as indicators for abundance, recruitment, and exploitation level and are used in the management of the fishery (Tarbath and Gardner, 2013). A change in abundance is indicated when both catches and CPUE decline or increase simultaneously. A decline in catch and CPUE together with a decrease in median length may be caused by low recruitment and/or low fishable biomass. Important management tools include TACs and minimum size limits. These are set for all zones separately due to high spatial heterogeneity in growth (Haddon and Helidoniotis, 2013). The MLS is set in relation to the estimated median size at 50% maturity. In 1985, the TAC was based on the average recent catch with an additional 10% bonus. Over the years, the TAC decreased, adjusted according to the stock status (Tarbath and Gardner, 2013). The TAC is set considering a range of inputs about the state of the fishery and evaluations from the industry and the government. Regional catch caps can be implemented where appropriate, and areas can be closed at short notice during the year.

A management strategy evaluation using an age-structured production model was developed for Patagonian toothfish (*Dissostichus eleginoides*) near sub-Antarctic Prince Edward Island by Brandão *et al.* (2002). Both CPUE and catch-at-length data were used as model input. A generic management procedure was developed to deal with situations where CPUE and length data give conflicting indications of stock status. Control rules to determine future TAC differed for each combination of CPUE and direction of the length data trend. Butterworth *et al.* (2010) conducted a management strategy evaluation comparing data-poor and data-rich scenarios by using two different control rules for the same stock – one in which the future TAC is altered according to the indicator mean length in the catch (considered to be a data-poor scenario) and another which makes use of commercial CPUE. For the same level of risk to the stock, the management plan for the data-poor situation results in lower average TACs with greater inter-annual variability than the data-rich scenario. The framework can support simulation testing of harvest control rules for data-poor fisheries (Butterworth *et al.*, 2010). Information on relationships of indicators (the trend in CPUE and mean length in the catch) to resource status and statistical properties of indicator measurements could be inferred from other fisheries (Brandão and Butterworth, 2009).

Punt *et al.* (2001a) used Monte-Carlo simulations to evaluate the performance of indicators for management of broadbill swordfish, *Xiphias gladius,* off Eastern Australia. In different scenarios of future effort trajectory, depletion status, and stock structure, it was found that indicators based on length or weight usually perform better than those depending on catch rate alone.

Southern and Eastern scalefish and shark fisheries in Australia (a multi-species, multi-gear fishery) use tier-based harvest control rules (Smith *et al.*, 2008). Depending on the level of available data per stock, different harvest control rules are applied. For data-limited stocks, quotas are set as a proportion of the average catch and according to trends in CPUE over the last four years. The fishery can be closed when catch rate drops below a limit reference point.

12 Discussion - Implications for the Development of an Indicator Approach for Scottish Shellfish Stocks

12.1 Indicators for *Nephrops* and Scallop Stocks

Scottish *Nephrops* and scallop stocks can be considered data-rich, because both fisheries-dependent and fisheries-independent data from scientific surveys are available. For *Nephrops*, underwater TV surveys are conducted annually as the basis for the stock assessment (ICES WGNSSK, 2015). *Nephrops* are a quota stocks with international TACs being set according to the estimated abundances. Individuals cannot be aged directly and assessments are length-based. Length distributions from surveys, commercial and discard sampling are available and CPUE or LPUE can be calculated.

For scallops, analytical assessments are performed for defined areas for which the available data are sufficient. In contrast to *Nephrops*, scallops can be aged using shell growth rings and age-length keys can be constructed to allow for an age-based assessment. The assessment makes use of both dredge survey indices and catch at age data from the fishery. Scottish scallop stocks are not subject to TAC regulations. There are no agreed reference points, and advice is given based on based on fishing mortality, abundance and recruitment trends (Dobby *et al.*, 2012). The analytical assessments may not be conducted at the spatial scale appropriate to inform local management and although the dredge survey does not cover all areas, it could potentially be used to provide abundance indicators (including recruitment) at a finer spatial scale.

For *Nephrops* and scallops, weights are currently derived using historical length-weight relationships and assumptions are made about the maturation schedule (Dobby *et al.*, 2012; ICES WGNEPH, 2013). These relationships could be updated using the data collected on surveys. A range of catch, catch-rate, length-based and spatial indicators can be calculated based on data available for these (data-rich) stocks. It would thus be possible to determine whether survey and stock assessment results can be corroborated by an indicator-based approach.

12.2 Indicators for Crab and Lobster Stocks

In contrast to scallop and *Nephrops*, Scottish crab and lobster stocks are considered data-limited. No surveys are conducted, and only fishery-dependent data (landings, length frequencies from commercial sampling) are available. As is typical for moulting animals, there is no direct age determination method available. The regional assessments of crab and lobster stocks, carried out by MSS, use landings and length frequency data and LCAs to estimate fishing mortality for males and females separately (Mesquita *et al.*, 2016). The Scottish stocks are not subject to TAC regulations, and apart from the Western Waters effort regime, there is no established management framework to control fishing mortality; the main regulatory tools are implemented through licensing and technical measures including MLS. The development of indicators and methods for data-limited stocks should therefore be considered for Scottish crab and lobster stocks (See examples in section 11).

The Data Collection Framework (DCF) of the European Union requires the collection of some fisheries data, such as effort, landings, as well as length frequencies from commercial sampling. Data on individual weights and maturity status (besides the number of berried females) are not routinely collected under the DCF. No routine discard sampling is done for Scottish creel fisheries. While discard mortality may be low, knowledge of catch composition can provide information on recruitment variability, stock dynamics and can be used for indicator development. Since 2009, some vessels in EU member countries have started carrying on-board video monitoring (CCTV), which allow analysis of catch composition to fully document activities in relation to cod (Needle *et al.*, 2015). The video monitoring may help with discard rate estimation and morphometric length inference. However, age, sex, weight and maturity cannot be monitored with CCTV and such data would also in the future need to be collected through sampling.

12.2.1 Size-based Indicators

We conclude that length-based indicators are the most suitable for the data-limited (Scottish) shellfish stocks, because they can be easily calculated and the required data are routinely collected. Data on catches and length frequencies can provide information on fishing mortality and selectivity and some indication of stock status. In contrast to scientific surveys, commercial fishing occurs in a non-standardised way, such that conclusions on stock status from these indicators should be drawn with caution.

Commercial catch sampling typically involves collection of length frequencies by sex and separately for berried females. The data on size at maturity and weight-at-length for particular species and assessment areas are not as easily available. Knowledge of the maturation schedule can be of importance as it relates directly to reproduction and potential recruitment. Therefore, it is important to update regularly estimates, such as L_{mat}, which are used in assessments or as reference points. Data on individual weight can help estimate yield and verify empirical length-weight relationships which are used to calculate biomasses and in models, such as production models. Individual weights can be used to calculate condition. Over time, changes in condition could thereby be detected and compared among different stocks of the same species.

For data-limited stocks, additional size-based data could be made available either during commercial sampling, self-sampling or through separate specific sampling programmes.

12.2.2 CPUE/LPUE and Spatial Indicators

CPUE or LPUE are also potential indicators - provided that both catch (or landing) and effort data are available. For larger vessels, fishing effort could potentially be estimated either using the number of days absent from port, AIS (Automatic Identification System), or though analysis of electronic monitoring systems (for example Vessel Monitoring Systems (VMS) for vessels >12 m, electronic logbooks for vessels >15 m). VMS data are routinely collected for vessels >12 m at a minimum required ping rate of 2 hours, which may not be sufficient to detect all fishing activity. While the calculation of effort in this manner may be possible for fisheries using trawl and dredge, it is less likely to provide a useful effort measure for creel fishery vessels without additional data on the numbers of creels or pots fished. This information is generally available only in the Shetland Islands where the numbers of creels or pots fished have to be detailed in the logbooks allowing for a calculation of effort and CPUE (Mesquita *et al.*, 2016). Such effort data would be desirable also for Scottish vessels in other areas. The extension of the use of electronic logbooks could further enhance data acquisition. A pilot project for collecting additional data for brown crabs west of Scotland from voluntary logbooks, GPS data and questionnaires was carried out (Anon., 2010). GPS loggers can deliver higher ping rates and allow better fishing effort calculation, but being voluntary may not cover vessels and areas representatively.

Spatial indicators can be calculated as well when the origin of catches or landings is recorded. In logbooks for vessels >10 m and for commercial sampling, data on the origin of landings is routinely recorded at the level of an ICES rectangle. From electronic logbooks or with help of VMS data, information at a higher resolution can be acquired (Bastardie *et al.*, 2010; Gerritsen and Lordan, 2011; Hintzen *et al.*, 2012). Scottish vessels <10 m declare weekly landings in weight per species and ICES rectangle on the Fish 1 form, which give some spatial resolution and allow for LPUE calculation.

12.3 Future Direction

In this review, we have explored a number of different indicators and related methods that have been applied in the management of data-limited stocks. Potential indicators relate to abundance, length frequency distributions, individual condition, mortality, reproductive characteristics, as well as spatial distribution of both stocks and fisheries. On the basis of the review, we consider that length-based indicators are likely to be most suitable to monitor changes in catch composition and may give information of fishing mortality and stock status for the data-limited Scottish crab and lobster stocks.

There is also scope to improve data availability and quality to assist the development of an indicator-based approach. To account for spatial and temporal differences in growth and maturation, estimated length-weight relationship and maturation size should be updated. Additional data on fishing effort and discards could deliver the basis for further development of indicators. If a programme to collect fishing effort data for creel/pot fisheries were established, CPUE or LPUE could be calculated and potentially developed as indicators. More intensive discard sampling would allow for an assessment of the number of undersized individuals, soft-shelled individuals and berried females in the catches. While length data are collected routinely, an assessment of coverage and sampling design is advised. It is crucial for the interpretation of derived indicators that the sampled data are representative of the fishery landings or catches from the respective assessment areas of each species. The development of an appropriate sampling design can be demanding when very diverse fisheries and species have to be covered and multiple measurements performed. Further development of electronic remote electronic monitoring offers the potential to support some aspects of data collection (Needle *et al.*, 2015).

Sampling often occurs at markets or processors and detailed information on the origin of the samples (ICES rectangle or at finer spatial scales) needs to be available. Self-sampling by the fishers could improve data quality for data-limited stocks, since measurements from unprocessed landings can be taken, and the composition (including discards), the origin and time of catch noted directly. However, the sampling process, with inclusion of all relevant size categories and random selection of individuals, can be demanding. If management measures such as MLS are in place, there may be an incentive for fishers not to sample catches below MLS. The development of self-sampling schemes at a larger scale would require a high degree of quality control to ensure accurate and unbiased data. Results of a pilot project for collecting additional data, effort and length frequencies, for brown crabs west of Scotland suggest that self-sampling or self-reporting can be feasible (Anon., 2010). As a result, it was suggested that the process could be facilitated in the future by establishing an assessment framework or a set of indicators beforehand and to allow for regular feedback between fishers and scientists (Anon., 2010). Within Scotland, a series of pilot projects to support sustainable Scottish inshore fisheries have been funded by the European Fisheries Fund, which include projects on self-reporting (EFF, www.seafish.org/research-economics/evidence-gathering-in-support-of-sustainable-scottish-inshore-fisheries). The results of a series of pilot projects indicate that self-reporting and fishers' knowledge can support improved stock assessment and management of data-limited stocks in Scottish inshore waters. These suggest that many of the data deficiencies could be addressed through a combination of self-sampling and electronic monitoring technology (Course *et al.*, 2015).

With the currently available data for lobster and crab stocks, areas can be identified where data are sufficient to support an indicator-based assessment. Logbook information allow for the calculation of catch- or landing-based indicators and spatial indicators of fisheries. To evaluate how indicator values are affected by changes in fishing regime and resulting changes in the underlying stock, simulation tests can be run. Based on this literature review, a next step for the work undertaken for ROAME SU0100 will be the development of length-based simulation models. These will be used to evaluate the effect of different stock-recruitment relationships and levels of natural mortality. Including sex structure in the models allows for sex-specific growth and fishing mortality. Indicator-based harvest control rules can be developed and simulation-tested in various management scenarios. Indicator-based assessment and management strategy evaluation will be further investigated as a tool for data-limited Scottish shellfish stocks.

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