

Reference points and reference directions for size-based indicators of community structure

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Size-based community and ecosystem metrics, such as mean body mass and the slopes of size spectra, have been proposed as indicators to support the Ecosystem Approach to Fisheries Management (EAFM). These metrics show relatively consistent responses to size-selective exploitation, and “unexploited” indicator reference points may be predicted with models of size-structured foodwebs. Whereas unexploited reference points provide a baseline for assessing the relative magnitude of fishing impacts, target or limit reference points are needed to guide management. Values for target or limit reference points are difficult to justify on scientific grounds. However, given that fishing impacts in most ecosystems need to be reduced to meet the objectives of the EAFM, we argue that reference directions provide alternative medium-term management targets. We show that the power of surveys to detect trends consistent with reference directions depends on the range of body size classes included in the analysis. Selection of different size ranges will weight metrics to respond to the release of small fish from predation, the depletion of larger individuals as a consequence of exploitation, or both. Such weightings may not be consistent over time, because the differential vulnerability of larger species, within-population changes, predator–prey relationships, and the effects of competition depend on contemporary rates of fishing mortality and the history of exploitation. The power of the surveys investigated is poor on time scales of less than 5–10 years. Therefore, size-based indicators provide better support for medium-term, rather than year-on-year, management decision making.

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Introduction

Size-based community metrics have been proposed as indicators to underpin the Ecosystem Approach to Fisheries Management (EAFM). Their intended use is to monitor ecological change and to assess the performance of management (Shin *et al.*, 2005). Size-based metrics respond to fishing impacts because body size determines the vulnerability of individuals, populations, and communities (Gislason, 2002). Fishing-induced changes in size-based metrics can be attributed to the differential vulnerability of larger species, within-population changes in size composition and life history, genetic changes, predator–prey relationships, and the effects of competition. Therefore, trends in these metrics reflect the overall

response of the community to mortality. A principal attraction of size-based metrics is the widespread availability of species-size-abundance data collected during ongoing monitoring programmes (Rochet and Trenkel, 2003).

For indicators to support decision-making, managers need to know the values associated with specific ecosystem states (Rice, 2003). These values are known as reference points (Sainsbury and Sumaila, 2003). For contaminants, reference points may be set to zero, or to the lowest detectable concentration, reflecting a wish to remove harmful substances that provide no ecological, social, or economic benefits from the marine environment. For target populations, reference points may be set to ensure that society accrues the ecological, social, or economic

benefits that fisheries provide, without compromising sustainability.

Reference points that relate to the unexploited situation may be appropriate for assessing the overall impact of fishing, because their use avoids the “shifting baseline syndrome”, where baselines set with a short-term perspective represent an increasingly impacted state over time (Pauly, 1995). However, this does not imply that the management objective is the unexploited state, because society deems some impacts acceptable, given the benefits that fisheries provide. Ultimately, setting a management objective is a societal issue, though science can provide commentary on the consequences of setting different objectives, and how to meet them (ICES, 2003).

Reference points that might support ecosystem-based management (Figure 1) include those for the unexploited ecosystem (or component), target reference points associated with the favoured state of the ecosystem (as a trade-off between environmental, social, and economic benefits), and limit reference points which, if exceeded, indicate that the ecosystem will be subject to serious or irreversible harm (ICES, 2001). As estimates of indicators contain measurement error, precautionary reference points may be used to guarantee a high (preferably specified) probability that the limit reference point is not exceeded. Indicators must be assessed regularly in relation to reference points, to identify changes in status of the system.

The aim here is to assess whether reference points can be developed for size-based indicators, and to suggest how size-based indicators might be used to support the EAFM if reference points cannot be established. We assume that a principal objective of the EAFM is to minimize ecosystem impacts associated with the capture of target species (FAO, 2003).

Reference points

A reference point for the unexploited state (URP) for size-based indicators might be based on historic data, or on models of the structure of unexploited communities. Historic data are unlikely to help describe the unexploited state, because exploitation has usually preceded scientific investigation, and the environment, as well as fishing intensity, has changed over time. Therefore, in practice,

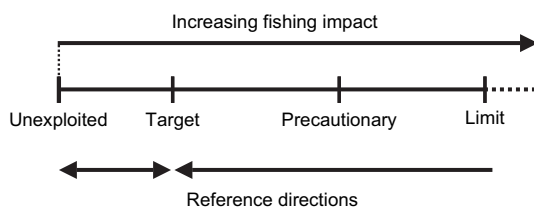


Figure 1. The relationship between reference points and reference directions for an indicator of fishing impact.

URPs have to be based on predictions of the structure of fish communities subject to contemporary climatic influences in the absence of fishing.

Numerous theoretical and empirical analyses consider the processes controlling size distributions in aquatic communities and foodwebs. Size distributions (in length or weight) are typically described using size spectra, also known as abundance–size relationships (Brown and West, 2000; Kerr and Dickie, 2001). Various models have been developed to predict the slopes of size spectra, either for all organisms in a foodweb (Kerr, 1974; Dickie, 1976; Benoit and Rochet, 2004), or for subsets of those organisms (Gislason and Lassen, 1997; Shin and Cury, 2001). Such slopes are sensitive to fishing impacts (Duplisea and Kerr, 1995; Rice and Gislason, 1996), and have been considered as indicators to support the EAFM (Rochet and Trenkel, 2003; Trenkel and Rochet, 2003).

A URP might be derived from the slope in an unexploited system, and Jennings and Blanchard (2004) demonstrated a method, based on macroecological theory, for predicting such a reference point from estimates of transfer efficiency (TE) and predator–prey body mass ratios (PPMR). They applied the method to North Sea data, and compared the contemporary fish community with the modelled community in the absence of exploitation. Size and trophic structure of the fish community were described from trawl survey data, PPMR was calculated from the slope of the relationship between trophic level (TL, estimated using nitrogen stable isotope analysis), and body mass for the fish caught, and a range of TE estimates was applied, consistent with those reported in other marine ecosystems (Ware, 2000). There were clear differences between modelled slope for the unexploited community (Table 1) and the contemporary (2001) slope of -1 . The slope of the unexploited size spectrum can also be used to calculate a URP for mean body mass (Table 1). Mean individual body mass in the 2001 community (size range of individuals included 64 g–66 kg) was 144 g, 38% of the value expected in the absence of exploitation (when $TE = 0.125$).

Table 1. Predicted mean body mass of fish (g) in an unexploited North Sea, for various combinations of body mass range (x –4096 g) and transfer efficiency (TE), and the associated slopes of the unexploited size spectrum that apply across all body mass classes, based on the assumption of constant predator–prey mass ratio (PPMR) and TE.

TE	Mean body mass					Slope
	$x = 16$	$x = 32$	$x = 64$	$x = 128$	$x = 256$	
0.100	83	155	284	512	913	-0.136
0.125	89	164	297	532	941	-0.100
0.150	95	172	310	551	966	-0.068

Although URP for size spectra and mean body mass can be calculated, the approach is often difficult to apply because the model applies to all animals present in each size class, not to a specific subset. Because fishing gears used during monitoring surveys are highly selective, the size composition of catches differs from the size composition of the community. Catch data would therefore need to be corrected for catchability and availability, to allow size compositions to be compared with URP. Few estimates of catchability and availability exist for non-commercial species (Sparholt, 1990; Harley *et al.*, 2001), and most survey trawls do not quantitatively sample invertebrates and small pelagic fish that dominate biomass in the lower classes of body mass (Figure 2; Jennings *et al.*, 2002b). Limit (LRP) and precautionary reference points (PRP) for size-based indicators have proven difficult to develop and justify. Despite numerous compilations of size spectra for marine fish communities (Bianchi *et al.*, 2000), there is no evidence that a specific slope value would reflect a limit beyond which serious or irreversible harm might occur, largely because slopes are gear- and area-specific (Daan *et al.*, 2003). Consequently, precautionary reference points are not easily justified.

Justifying a TRP is also difficult, because its selection is driven by society, and yet, apart from a desire to avoid the loss of genetic and species diversity (WSSD, 2002; Dulvy *et al.*, 2003), society has not expressed consistently strong views on how fish communities should be structured. Despite a lack of theory to underpin the selection of LRPs and TRPs for size-based indicators, RPs could be selected on less stringent criteria, to provide immediate targets for managers. However, we consider that substantive challenges to RPs are unlikely to be upheld by managers if it cannot be demonstrated that the LRP is associated with serious or irreversible harm.

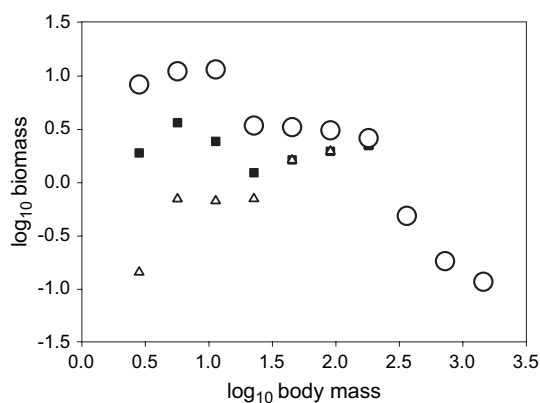


Figure 2. A complete size spectrum for animals of 2–1024 g in one region of the central North Sea (circles), and size spectra for the same region as they would appear if small pelagic fish (triangles) or invertebrates (squares) that are not sampled by the standard groundfish survey trawl were excluded (data from Jennings *et al.*, 2002b).

Reference directions for trends

An effective EAFM should minimize the ecosystem effects of fishing, while meeting management objectives for target species (FAO, 2003). To support the EAFM, the relative effects of fishing on fish communities can be measured with size-based indicators.

Although RPs for size-based indicators are difficult to develop and justify, a community with size structure closer to the unexploited state will, by definition, be less impacted by fishing. Therefore, these indicators can be used to assess whether target stock management strategies are having positive or negative effects on fish communities. Because the indicators respond to direct and indirect fishing impacts, possibly with considerable time-lags, there would be many potential causes of observed trends, including prey release, loss of large species and individuals, or genetic selection. Undesirable trends should be a stimulus to identify the causes and to initiate remedial management action. Therefore, the principle role for size-based indicators would be to provide surveillance of the status of the fish community.

One potential application of size-based indicators in supporting the EAFM is summarized in Figure 3. When the status of target populations is improving in response to management action, and the status of the fish community is also improving (A), the management action is apparently consistent with the EAFM. Note that the reference to the status of the target populations improving or deteriorating refers to an increasing or decreasing proportion of all fished populations being within safe biological limits (or other measures of sustainability), and not to the status of individual populations. When the status of target populations deteriorates, but the status of the fish community does

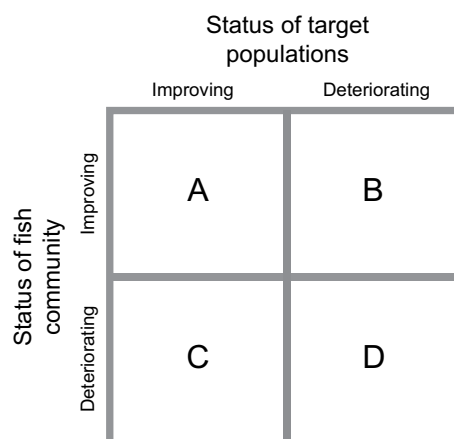


Figure 3. Relationships between an indicator of the status of target populations (e.g. proportion of all target stocks in a region exploited within safe biological limits), and an indicator of the status of the fish community (e.g. mean body mass) that support the implementation of an ecosystem approach to fisheries management.

not (B), then management action should focus on target populations. When the status of target populations is improving, but the status of the fish community continues to deteriorate (C), then ecosystem considerations have not been addressed adequately. When the status of both target populations and the fish community deteriorate (D), management is failing to address all relevant objectives.

Therefore, to support the management decision-making process, knowledge of the direction of trend in the indicators can be sufficient, and there may be no absolute need to identify reference points. Given that the structure of exploited fish communities in many ecosystems is currently driven by fishing intensities that have reduced a number of target populations to below sustainable levels (FAO, 2002), trends in size-based indicators consistent with reduced fishing impacts will provide appropriate guidelines for managers in the medium to long term (over decades).

For simplicity of presentation, the illustration in Figure 3 does not identify management actions when there is no change in the size-based metric, or in the proportion of target populations within safe biological limits. Nor does it identify changes in the values of other indicators that would be required to support the EAFM. If there was no change in the proportion of target populations within safe biological limits during any reporting period, and if this proportion was <100%, then the management objective of improving the status of target populations remains. Similarly, if there is no change in the size-based indicator, the management objective is still improvement, consistent with the EAFM. The time scale on which assessments of the target populations and fish community are conducted should be consistent with the statistical power of the surveys to provide information on trends. Managers risk squandering credibility and resources if they respond to noise rather than to signal (Rice, 2003).

When working with reference directions, effects of gear selectivity are less important than when working with absolute RPs, but it must still be assumed that the direction of trend in the size structure of the samples reflects the direction of trend in the community (Link *et al.*, 2002; Trenkel and Rochet, 2003). When the indicator is calculated from data for size classes and species well sampled by the gear, then this assumption is likely to be acceptable, but in most cases, a lower limit must be set to the size range from which the indicator is estimated, and the metric should be applied to a selection of well-sampled species, to reduce noise and to increase signal in the time-series.

Power to detect trends

If reference directions are adopted as management targets, it is important to know the time scales during which significant trends might be detected. To an extent, this question can be answered using statistical power analysis, although its principal weakness is the dependence of the

variance estimates on a statistical rather than an ecological model. In practice, this weakness has a small effect on the significance of power calculations, because consistent year-on-year trends would be a feature of the reference directions used to judge the success of management. Therefore, power analyses are assessing power to detect societal expectations, as well as ecological processes.

As an example, we consider the properties of size-based indicators calculated from North Sea trawl survey data, and the power of these to detect reference directions. We emphasize the interaction between the size range used in the calculations, the retrospective trend in the metric, the interannual variance in the metric, and the power to detect trends in future years.

The size range used will influence trend and variance, because survey gears do not sample all size classes with equal efficiency, and because large and small size classes respond differently to fishing. Larger size classes will be depleted by the direct effects of fishing, whereas smaller size classes may proliferate because predators have been depleted (Daan *et al.*, 2003; Dulvy *et al.*, 2004). Therefore, an indicator biased towards larger fish will reflect the direct effects of fishing more strongly than an indicator based on data for smaller fish.

To investigate how the selection of different size ranges could affect the power of a survey to detect trends in a reference direction, we used data from the North Sea International Bottom-Trawl Survey (IBTS). Community metrics were calculated from species-size-abundance data (number per hour fishing) for 107 rectangles sampled every year from 1982 to 2000. Mean values of the metrics were then calculated among rectangles within years. Further details of the IBTS data and metric calculations are provided in Jennings *et al.* (2002a), and Nicholson and Jennings (2004).

Mean body mass of an individual in the catch was calculated as the weight of the catch divided by the number of individuals. The slopes of biomass size spectra were calculated by assigning individuals to \log_2 body mass classes, and estimating total biomass by class. The linear relationship between \log_{10} body mass midpoint of \log_2 body mass class (x) and \log_{10} biomass (y) was described as $y = bx + a$, where b is the slope of the size spectrum. Mean maximum mass of mature fish was calculated according to Nicholson and Jennings (2004), but here only mature fish, defined as those longer than the estimated length at 50% maturity (L_{mat}), were included in this metric, so that it reflected the composition in the mature component of the fish community. However, it is accepted that this approach is subject to some error when the L_{mat} of many North Sea species is only measured occasionally and/or in small areas of the North Sea. Following others (Shin *et al.*, 2005), mean maximum mass of mature fish is considered a size-based metric. However, it is sensitive to the relative abundance of species in the community, rather than to their realized size.

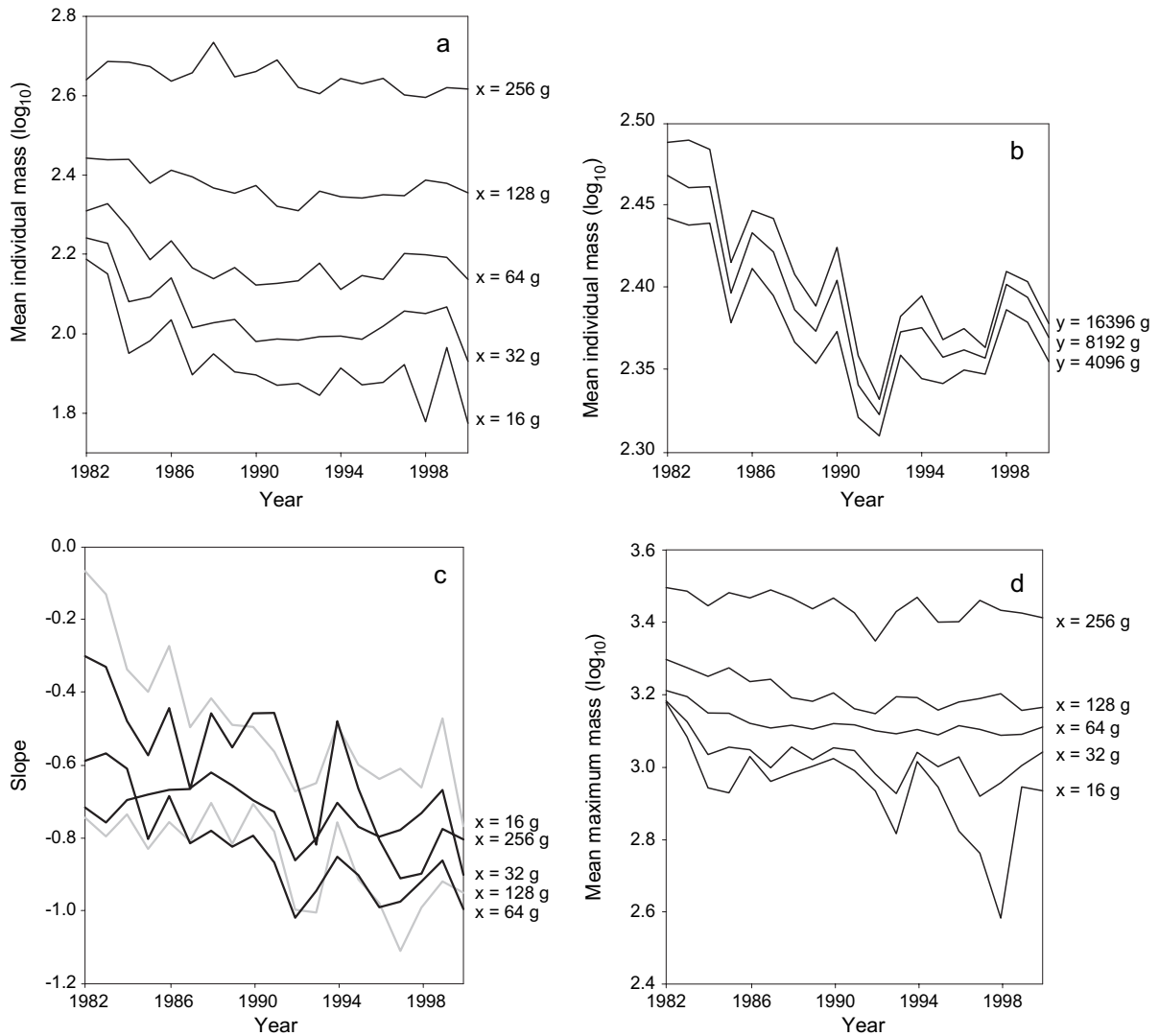


Figure 4. North Sea demersal fish (IBTS data): temporal trends in (a, b) mean body mass, (c) slope of size spectrum, and (d) mean maximum body mass of the mature component of the community, as a function of (a, c, d), the minimum body mass class (x) included in the analysis (range $x=4096$ g), and (b) the maximum body class mass (y) included in the analysis (range 128 g– y).

We calculated trends in the metrics within the size range $x-4096$ g, where x varied from 16 to 256 g. Mean individual mass decreased faster over time when smaller mass classes were included, but also interannual variance appeared to be larger (Figure 4a). Changing the upper size class, while holding the lower size class at 128 g, did not have such marked effects on trends in the metric (Figure 4b). Trends in slopes were also dependent on the range of size classes included in the estimate, but the patterns were less consistently related to the choice of minimum size (Figure 4c). For mean maximum mass of mature fish, interannual variance was highest when the smallest or largest size classes were used in the calculation (Figure 4d).

As well as on magnitude and pattern of trend, sampling scheme, and significance level of the test, power depends on the residual variance, ψ^2 (Nicholson and Fryer, 1992). To identify the size range that optimized power, and hence the trade-off between trend and variance, we calculated the power to detect future trends in the community metrics, based on the difference-based variance estimation method recommended by Gasser *et al.* (1986), and following the approach of Fryer and Nicholson (1993). When the projected trend in slope, b , is $\neq 0$, then power depends on the number of data points in the time-series, and the signal to noise ratio b/ψ . Projected trends in slope were assumed to correspond to the recorded trends (1982–2000).

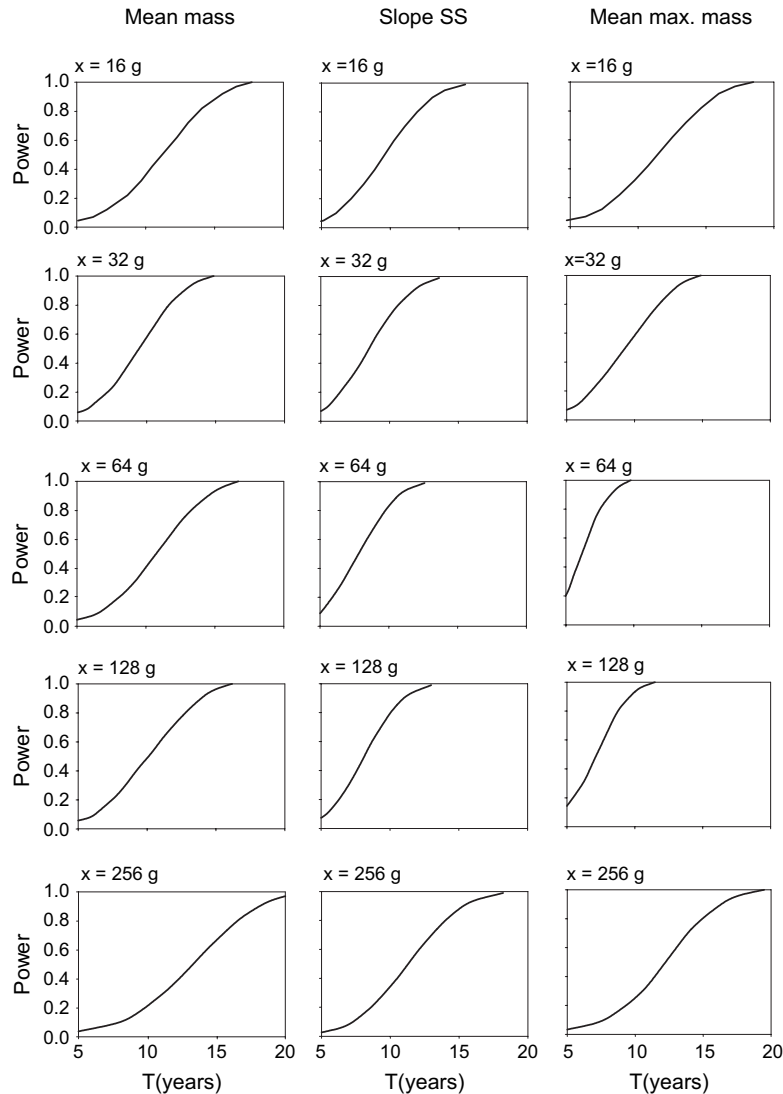


Figure 5. Power to detect future temporal trends (based on observations for the years 1982–2000) in the mean mass, slope of size spectrum, and mean maximum mass of the mature component of the North Sea demersal fish community, as a function of the number of years for which data are available, and of the minimum body mass class (x) included in the analysis (range x –4096 g).

The calculations (Figure 5) show that metrics calculated from a range starting with intermediate body mass classes provide the greatest power to detect future trends. For mean mass, the power is maximized when $x = 32$ g, whereas for the slope of the body size spectrum and mean maximum mass of mature fish, power is maximized at $x = 64$ g. The size window that provides the greatest power to detect fishing effects may be expected to change with the history of fishery development. In a new fishery, larger size classes should be depleted more rapidly at first, but progressively smaller size classes may show the greatest response to fishing as effort continues to increase, if only because of higher noise/signal ratio, as the larger fish become more

rare. In the power analyses, it was assumed that the estimate of variance from retrospective analysis applies in future years, an assumption that will be violated when fishing mortality and/or the environment change. Therefore, when fishing mortality rises, the age and size structure of the population become more sensitive to annual recruitment events, leading to greater relative changes in abundance. Conversely, when fishing mortality falls and abundance rises, interannual variance in abundance is likely to fall, because effects of annual variations in recruitment will be buffered by the increasingly extended age structure of the population, possibly augmented with effects of density dependence. Consequently, true power will be lower than

predicted when abundance is declining, and higher than predicted when abundance is increasing.

Conclusions

Practical issues currently preclude the development and adoption of firm reference points for size-based indicators. However, an appropriate target to support the EAFM would be a reference direction that is consistent with a decline in the overall human impacts of fishing on the community, and thereby on the ecosystem.

Size-based metrics would better support medium-term rather than year-on-year management evaluation, because (i) they are unlikely to be appropriate for detecting responses to management action on time scales <5 years, and (ii) the response to management action cannot be quantitatively decomposed in the contributing causal factors without extensive additional research. Over the medium-term, trends in the mean mass per individual, or the slope of the size spectrum, possibly coupled with an index of trends in the abundance of vulnerable species, provide a good broad assessment of the status of the fish community that can be used to identify trends in the status of fished ecosystems, and to provide a method for identifying and highlighting specific concerns.

Trends counter to the reference direction could be used to trigger analyses to identify causes of the undesirable trend and scope for relevant management action. Trends consistent with the reference direction suggest that management is consistent with the EAFM, but specific targeted research might still be needed. The power of surveys to detect trends consistent with a reference direction depends on the range of size classes included in the analysis. Selection of different size ranges will also weight the response of metrics to the release of small fish from predation, the depletion of larger individuals as a direct consequence of exploitation, or both. Such weightings will not be consistent over time.

Ideally, indicators should be calculated for size classes that are well selected by the gear. Much of the interannual variance in indicators may be attributable to sampling inefficiency, and to the relatively large effects of interannual recruitment variation, if poorly sampled small size classes are included. However, excluding all smaller sizes will also limit the response of size-based metrics to the indirect effects of fishing, because any release of small individuals from predation will reduce mean mass, and steepen the slope of the size spectrum. Similar arguments apply to mean maximum mass, where the inclusion of smaller sizes and hence species with a lower maximum mass, makes the index more sensitive to prey release. Owing to the range of gears used for monitoring, and their differences in selectivity, values of size-based metrics are survey-specific. To avoid misleading comparisons within and among areas and ecosystems, these metrics might more usefully be expressed as relative indices.

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