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INDECO

Development of Indicators of Environmental Performance of the Common Fisheries Policy

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Results of the investigation of modelling methods and indicators of marine ecosystem status

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Table of Contents

1   INTRODUCTION

1.1   INDECO objectives

The combined effect of fishing on target species, non-target species and habitats may result in major impacts on marine ecosystems. In 2002, the EU agreed to begin the implementation of an ecosystem based approach to fisheries (EBFM) management under the Common Fisheries Policy (CFP) with the result that there is now a requirement to recognise the relationship between fishing activity and the environment within a management context.

Indicators which reflect the effect of fishing on the ecosystem have the potential to be useful tools for tracking spatial and/or temporal changes at an ecosystem level and can also be used as a means of assessing progress towards the achievement of policy objectives. While many indicators have been proposed, few have been implemented. The aim of this project is to identify a set of useful generic indicators that can be applied at the EU level to measure and monitor the progress of an ecosystem approach to fisheries management under the CFP.

The objectives are to-:

- Achieve a coherent approach to indicator use in the implementation of the CFP

- Identify a set of indicators which measure the impact of fishing on the environment

- Identify how these indicators may be used to measure the effectiveness of fisheries management

- Identify data deficiencies and propose cost-effective remedies

- Propose ways to ensure that the agreed indicators are incorporated in the policy process

The INDECO work packages prior to WP5 identified indicator criteria and the indices that are available. The main objective of this package is to describe methods for modelling potentially useful generic indicators and how these indicators may be incorporated within a management framework to monitor and evaluate the environmental performance of the CFP.

When used in conjunction with baseline reference points, indicators can provide a useful tool for monitoring progress towards achieving management objectives. This Work Package will discuss how baseline reference points have been previously used in the management of marine systems and how they might be applied to manage fisheries under the CFP.

Due to the relatively short time period and limited resources available the aim of this work package is not to develop new approaches to the modelling of indicators but rather to propose some relatively simple models for the analysis of a set of generic indicators. The feasibility, sensitivity and general utility of these indicators will be discussed for various marine ecosystems of interest.

1.2 Objectives of this deliverable

This report summarises findings of previous deliverables (D9 and D17). Also included here is a description of alternative indices for investigation that have been highlighted over the course of INDECO meetings. A key objective of this deliverable is to make recommendations on the future application and research of indicators that may offer a basis with which to assess the CFP.

In summary the specific objectives of this report are to-:

- Identify those generic indicators that by virtue of their general utility, acceptability, economic feasibility and sensitivity have been previously shown to provide a measure of the impact of fishing on ecosystems, either at the single-species, population or community level

- Discuss how these indicators may be applied to operational models and these models relate to the impact of fishing on the environment

- Describe the advantages and limitations of models that describe candidate indicators

- Highlight the difference between a theoretical and validated model and discuss the importance of validating models

- Recommend models that can be used to predict the impact of fishing and changes in fishery management practice on the ecosystem

- To discuss how the output of the operating model may be used in conjunction with baseline reference points to provide management information

2 INDICATOR CONCEPTS

2.1 Defining a measure to be indicated

Before embarking on a description of the potential limitations of the candidate indicators below, it is instructive to separate two concepts: measurement and indication. A measure directly quantifies the variable of interest (*eg.* macrofaunal species richness as a measure of biodiversity, biomass as a measure of fish population size). An indicator (*eg.* presence of an indicator species, catch-per-unit-effort) is defined as a surrogate of a measure that is too difficult, inconvenient or expensive to apply routinely (Landres *et al.* 1988, McGeoch 1998, Rolstad *et al.* 2002).

Where a measurable definition of a variable of interest is provided (*eg.* model simulated biomass, sampled macrofauna species richness) the relationship of any number of indicators with the measure can be tested. In this application, no measurable definitions of the 'success of the CFP' or 'ecosystem status' are identified. Instead, a number of indicators are proposed that are themselves assumed to be a measure of the variable of interest.

The inherent problem with this approach is that these indicators cannot be selected on the basis of their relationship with a common target measure (*eg.* model predicted biomass, macrofaunal species richness). Importantly, any fluctuations or trends cannot be validated or interpreted on a scientific basis beyond the index itself. Where a target variable such as fish population abundance is identified, without simulation or experimental testing, the links with an indicator, such as average size, remains hypothetical. For example, where significant reductions in the average size of fish are

detected, the only hypothesis that is falsified is that size of fish is constant or increasing. Further interpretations of such an indicator are dependent on untested assumptions of varying magnitude. For example, in the context of a fisheries indicator, the assumption that the average size of a fish is related to the health of the stock (a measurable definition could be biomass relative to biomass at 'maximum sustainable yield') may be valid under certain circumstances. Broader interpretations in terms of 'ecosystem status' and other related and undefined variables such as 'biodiversity' and 'ecological integrity' are reliant on much larger assumptions that are ecologically questionable. Importantly, in the former example, the 'biomass relative to biomass at maximum sustainable yield' is defined and measurable, and its relationship with size can be assessed in a simulation evaluation.

Practical problems may also arise where many measures are identified (*eg*. fish population abundance, species richness) to assess a broader target variable (*eg*. 'ecosystem status') it may be difficult to weight the relative importance of the composite measures (*i.e.* decide on an appropriate measurable definition of 'ecosystem status').

2.2   Simulation and experimental evaluation versus descriptive statistics

Since indicators are used to quantify a variable of interest that is usually dynamic, a minimum requirement of an indicator is that either it responds to changes in the variable of interest or the variable of interest responds to changes in the indicator. Consequently, the hypothesised relationship of an indicator with such a variable is a functional one. Such a relationship cannot be tested using solely descriptive data. Consider the example of a freshwater outfall that discharges into an estuary. Away from the outfall, the concentration of many different contaminants decreases and species richness increases. Hypothetically, the loss in species richness may be attributable to the increase in any single contaminant (*eg*. PCBs, heavy metals, nitrates) or combination of contaminants. Under these circumstances, an excellent negative correlation (that is coincidental) could be erroneously used to identify a predictive indicator of species richness. In this example, species richness was unrelated to contamination but rather the unobserved increase in variability in salinity nearer to the outfall.

Another important problem with using a predictive indicator is that the variable of interest should not be readily affected by other factors. Under such circumstances the target variable could change dramatically with no concurrent change to the indicator. Additionally, the correlation (negative or positive) among indices may be high but this should not be interpreted as greater support for changes in the variable of interest, they may be functionally unrelated to one another and the variable of interest.

Simulation or manipulative experimentation offer a solution to this problem. A fishery, for example, may be simulated and the relationship may be assessed between average length of fish caught and the model predicted stock biomass. Once a level of stock biomass that is undesirable is decided on (a trigger point, see below), the corresponding average length of caught fish may be calculated. In this case, the functional relationship is modelled and uncertainties over this process may be accounted for in assessment of the indicator. Under many circumstances the natural system may be too complex to be modelled. A manipulative experiment where either indicators are altered and the response in the target variable is assessed (in the case of a predictive indicator) or vice versa (a responsive indicator) provides a rigorous

means of testing the hypothesised functional relationship. Again, simulation analysis or manipulative experimentation require a measurable definition of the variable of interest and probably also a trigger point for management purposes.

An alternative approach (of which AMBI is an example; Borja *et al.* 2000) involves determining areas that are subject to a range of pressures and evaluating the response in the indicator. In some cases, 'pressure' may not have a measurable definition and the quantitative relationship between pressure and the index may not be exactly known. However, the index may be monitored before and after the advent of a source of 'pressure' and the response analysed statistically. Other 'reference condition' based indicators such as the River Invertebrate Prediction and Classification System (RIVPACS) used in England (Wright *et al.* 2000) , the Australian River Assessment System (AusRivAS) (Simpson and Norris 2000) and the Benthic Assessment of Sediment (BEAST) used in Canada have proven useful in freshwater systems. The only potential shortcoming of such methods is that of intrinsic temporal and spatial variability in the index. Extrapolations of the relationship between the index and the target variable to other locations may be incorrectly interpreted in terms of 'pressure' or 'disturbance'. Equally, where the methods are applied descriptively, the temporal sensitivity of the index to changes in disturbance is not established. It is possible that responses in the community occur many years after contamination, in such cases, the indicator may not respond over a temporal scale that is relevant to management.

2.3 Selection bias

Suppose that the temporal trend in the abundance of a fish population was known and compared with 1000 unrelated indices. Where alpha (rate of Type I error, the probability of a false positive) is set to 5 percent, on average 20 of these 1000 indices may be expected to show a similar correlation with abundance. In cases where many indices (for example the catch of 100 fish species) are under analysis, a certain proportion will offer a significant trend that is a product of sampling error rather than a significant relationship.

3 MODELLING WORKSHOP

After a review of the commonly applied methods and a practical workshop on time series methods and statistical power, ARIMA (auto regressive integrated moving average), polynomial and power analyses were applied to average weight and length data from the IBTS dataset. Attendees of the workshop applied the methods to specific indicators. For example, ARIMA modelling was investigated to describe the abundance indices of the annual EVHOE bottom trawl survey (Marie-Joëlle Rochet, IFREMER Nantes, France). Greek trawl fleet data were also analysed using generalised linear modelling to extract time effects that could be subject to trend analysis (John Harabalous, HCMR, Greece). Also investigated were benthic community biodiversity indices (AMBI) that were calculated for two locations in the Bay of Biscay (Iñigo Muxika and Angel Borja, Basque Country).

3.1 IBTS survey data. Dr Murdoch McAllister, Imperial College, UK

ARIMA models were investigated that could describe the available weight and length indicator time series. Since ARIMA models may involve many combinations of auto regressive and moving average components, a large number of prospective models were available to explain the data. Unfortunately, a time series in excess of 50 years are generally required in order to choose among ARIMA models using methods such

as AIC (Akaike's Information Criterion). Given that less than 30 years of data were available for most indices under investigation here, only simple ARIMA models were open to selection and in most examples there were no strong statistical grounds with which to select one model over another.

Other time series methods that require shorter time series such as polynomial modelling were also investigated and models of varying complexity were subject to selection. When indicators were normalised such that the trends could be compared, significant temporal covariation was detected (unsurprisingly since length and weight are closely functionally related). Statistical models were applied that could undertake time series analysis on the common temporal trend among length and weight indices.

An important criterion of an indicator is that important trends are detectable over a time scale in which remedial management actions may be taken. Statistical power represents the ability to correctly identify a significant trend in an indicator time series. Under power analysis, the size based IBTS trends could not achieve sufficient power (80%) for durations less than 12 years. A similar phenomenon was expressed in the abundance indices of the Bay of Biscay, where at least 10 years are required before a significant trend can be detected in 80% of cases. Unfortunately, in these cases these temporal scales are too large. In most management situations, a significant trend should be detected in 5 years or less. Even where power is maximised by assuming a one tailed hypothesis, neither length or weight indices offer a means with which to detect significant trends over time scales relevant to management.

3.2 EVHOE bottom trawl survey. Dr Rochet, IFREMER, France

ARIMA models of varying complexity were fitted to the abundance indices of 58 species from the EVHOE bottom trawl survey, 1987-2004. The 'best' model in each case was selected by AIC. Sufficient power (> 80 %) to detect a significant slope over a five year period was detectable in four abundance indices ('CAPOAPE', 'GADIARG','GALUMEL', 'LIZARAM'). Multivariate time series analysis provided a significant increase in power over component abundance indices. However, the multivariate methods still did not provide a basis with which to detect important trends over time periods that are appropriate to management.

3.3 Greek trawl data. Dr Harabalous, HCMR, Greece

Community level length and size data from Mediterranean bottom trawl surveys in Greek waters were analysed for significant temporal trends. Unfortunately, only a relatively short time series of data were available and regression analyses were inconclusive for average length and weight in addition to indices of biodiversity. Shallow slopes coupled with a relatively large degree of temporal variability lead to insufficient statistical power with which to detect significant trends.

3.4 AMBI. Bay of Biscay. Dr Muxika and Dr Borja, Basque Country

The soft sediment benthic community was sampled at two locations in the Bay of Biscay and a biotic index (AMBI, Borja *et al*. 2000) calculated. The sampling location subjected to previous (non-fishing) pressures showed a significant negative trend in AMBI, thus indicating recovery. Additionally, the relationship was sufficiently precise to offer reasonable statistical power over a six year time scale. In this instance the sensitivity of the indicator was assessed only in terms of recovery

from pressures unrelated to fishing. The indicator provided no significant trend away from low calculated values in the second location where relatively few pressures where observed.

4   CANDIDATE INDICATORS FOR MODELLING FISHING IMPACTS

The choice of indicator or metric for the detection of fishing impacts is critical and is dependent on the objective of the study, the specificity of the study and the scale of the investigation (Chabanet *et al.* 2005). Most indicators of fishing impacts are non-specific, i.e. they can detect a temporal and/or spatial change in a signal without identifying the actual cause of the change. A major weakness of most indicators is the difficulty in distinguishing the response to fishing impacts from other ecological and biological stressors that can evoke a response in the indicator (*eg.* environmental forcing or pollution events). It is important that indicators have a demonstrable causal link to the impact but such a link is difficult to identify, particularly for community level indicators (Keough & Quinn 1991). Previous studies have shown that the use of a set of metrics provides more information than a single indicator considered in isolation (ICES 2005). For this reason, a hierarchical concept of indicators (Chabanet *et al.* 2005) or 'holistic suite' of indicators (Rochet & Trenkel 2005) which apply to different levels of organisation (species, population, community and region) may provide the best available method of identifying, monitoring and forecasting the impact of fishing on the ecosystem.

There are probably as many different indicators as there are objectives. The INDECO working group identified a number of common indicators which were considered to be the most 'useful' according to the criteria of Rice and Rochet (2005). Four of these indicators with a high level of perceived utility are catch, catch-per-unit-effort, effort and size spectra. A brief discussion of how these indicators may be incorporated in models to assess the impact of fishing follows.

4.1   Catch

A direct estimate of stock abundance is a simple but powerful indicator of state. When direct estimates are unavailable or too costly to achieve, catch or catch-per-unit-effort may provide an alternative indirect measure of abundance.

Catch has been used as temporary indicators of stock status in established fisheries while other indicators are developed for implementation (Scandol & Forrest 2001) and in developing fisheries where information is scarce (Degnbol 2001). Information on total catch is usually collected as a matter of routine for most commercial fisheries and the metric requires no standardisation or subsequent processing (*eg.* standardisation) and is therefore considered to be 'transparent' with a high degree of acceptance among stakeholders. Catch can be considered a measure of productivity and a possible indicator of characteristics of the fish population. For example, models may be projected and the future status of the stock assessed with respect to varying levels of total allowable catch. If current catch levels are projected, a fishery may be assessed in terms of sustainability (a measurable quantity could be the number of years until the biomass is some fraction of that at maximum sustainable yield).

However, there are difficulties associated with the interpretation of catch as an indicator of stock size. Primarily, catch is affected by many other factors such as effort (targeted and untargeted) and market price; all of which are unrelated to stock size. Catch is likely to be closely affected by regulatory measures such as time-area

closures and quotas. In isolation, catch is unlikely to be strongly related to population size at levels that are not heavily exploited. In fisheries in which there is high exploitation and the market value of the fish is high, a possible trigger point could be the fraction of the quota not met.

It is possible to set trigger points for the detection of recruitment or survival failure of the standing stock. However, these trigger points can result in a high level of false positive outcomes (see Scandol and Forrest 2001). Prior to using catch as an indicator of stock status it is advisable to decide on appropriate reference points, the magnitude of the signal that warrants management intervention and the acceptable probability of false positive (and negative) outcomes in consultation with stakeholders.

4.2 Catch-per-unit-effort

Of those indices based on a historical time series of commercial or survey data , catch-per-unit-effort (CPUE) may be considered the most likely to contain information of relative abundance over time. CPUE have been applied widely in fisheries stock assessment and a range of models of varying complexity may be used to estimate population abundance, and reference points (*eg.* harvest rate at maximum sustainable yield, biomass relative to carrying capacity) (Hilborn and Walters, 1992). The relative merits and pitfalls of these methods have been illustrated in previous case studies (Harley *et al.* 2001, Punt *et al.* 2001) and the techniques are in a continuing state of review.

Much of the discussion to date has focussed on using single species models to provide information on the effect of fishing on a target species. Despite its recognized limitations (Paloheimo & Dickie 1964, Clark & Mangel 1979, Cooke, 1985, Mangel & Beder 1985, Gillis *et al*. 1993, Gillis & Peterman 1998, Walters 2003), catch-per-unit-effort (CPUE) is routinely used in stock assessments as index of relative abundance and trends in CPUE are considered to reflect trends in the relative abundance of fish populations (Hinton & Maunder 2004).

While trends in CPUE/Biomass provide a measure of changes in the relative abundance of that component of the ecosystem that is the single stock, an assumption that trends in CPUE of a single stock reflect the response to fisheries exploitation at the ecosystem-level is fundamentally flawed (Maunder *et al.* 2006). Aggregated CPUE has also been shown to be a poor indicator of the impact of fishing at either the ecosystem or community-level (Lorenzen *et al.* 2006). However, in many situations fishery catch and effort data is often the only information available which may provide an indication of the impact of fishing. Trends in a pressure indicator such as CPUE, when considered in relation to trends in other indices such as changes in mean species size or mean species length may provide insight into fishing impacts at an ecosystem level (ICES 2005).

A reference period (that is essentially chosen subjectively but should reflect a typical period of sustainable fishing) can be selected from a historic time series of CPUEs and then used to define trigger points. For example in the case of the Tasmanian scale fish fishery, an annual species trigger point is activated when the CPUE is less than 80% of the lowest CPUE observed during the reference period level 1994/95 to 1997/98 (ANZECC 2000)

4.3 Effort

The same advantages and disadvantages regarding the use of CPUE as an indicator of fishing impact also apply to the use of effort. The effort of fishery types such as trawling may be indicative of pressure on particular habitats such as soft sediments and deep sea corals reefs. The interpretation of the effort is however dependent on information of the likely distribution of the fishing activities.

There are two ways in which an indicator can provide a predictive estimate of the variable of interest. The response in the target variable to changes in the indicator may be assessed and the temporal discrepancy established. Possible examples could be the dredging of fishing lanes that leads to alterations to the physical environment that have known impacts on marine communities. The only possible indicator with this property is that of fishing effort data. Whilst it is likely that increasing fishing effort results in increasing disturbance in the marine environment, the quantitative relationship is not well understood.

An alternative approach involves constructing a model of the system (an operating model) that allows parameters to be 'fit' to observed data and projections to be made on the basis of the fitted model and parameters. As described above the latter method may be applied using the catch-per-unit-effort data that are available

4.4 Size spectra

Size spectra are common community level indicators of the impact of fishing on marine ecosystems (Pope *et al.* 1988, Bianchi *et al.* 2000) in which the relationship between the abundance or biomass of organisms and body size is commonly expressed as a linear regression but can also assume a parabolic form if the parameters of interest are expressed in logarithmic terms. Both the linear and parabolic forms of size-spectra models have inherent biases and linear slope and parabolic curvature can be influenced by factors other than the effect of fishing (*eg.* recruitment events) (Duplisea & Castonguay 2006)

However, size-spectra indicators have a strong theoretical basis and their response to fishing impacts has been validated in several systems (ICES 1998). Duplisea and Castonguay (2006) and Shin *et al.* (2005) suggest that since different size spectrum indicators reflect separate aspects of the community structure, a more effective approach is to consider a number of complimentary size spectrum indicators in order to assess temporal trends which provide a more comprehensive measure of the impact of fishing on the community.

Similar to catch and effort data, size may be included in an age-structured model and analysed with respect to varying catch and effort. Fish size data may provide information over the stock status but it may be the case that such an index is only sufficiently sensitive at relatively low stock sizes and high exploitation rates.

The use of reference points with size spectra metrics is still in development (Link 2005).

Table 1   A summary table of candidate indices, their possible target variables, advantages and limitations.

Index (class)	Target Variable	Advantages	Limitations
Catch-per-unit-effort (fish population indicator)	Abundance. Population density. Catchability.	Widely used in fisheries stock assessment. Well documented successes and failures.	Has been associated with previous stock collapses such as northern Cod and North Sea herring. Interpretation in terms of abundance may be confounded by catchability or shoaling effects.
Size, length or weight (fish population indicator)	Exploitation of the fish stock. Size selectivity of fishery. Size attrition due to over fishing.	Functionally related to population characteristics such as age. May offer a warning of high exploitation rates.	Average size is not considerate of the magnitude of catch. Significant reductions may occur only at already heavily exploited levels. May be compromised for populations with sporadic recruitment events and/or strong cohort structure. Where indices are not species specific, the interpretation of the index is likely to be confounded by changes to the target species
Fishing effort (pressure indicator)	Predictive indicator of stock levels	Where fishery type is available, may be interpreted in terms of disturbance to different habitats (eg. trawl fishing and marine benthos). Disturbance is likely to increase with increasing effort	At levels that are low relative to over exploitation changes in effort may be inconsequential. At low population abundance levels economic circumstances can allow effort to remain high while a stock collapses. Interpretation requires information of the type of fishery (*eg.* trawl) and may be improved by knowledge of the spatio-temporal distribution of effort. The exact relationship between effort and disturbance is not established.
Catch (socio-economic indicator)	Productivity	At given effort levels, related to the population abundance	Likely to be strongly dependent on fishing effort. Dependent on both the population abundance and demand. Dependent also on management decisions such as quotas and time-area closures.

5   CONCLUSIONS AND RECOMMENDATIONS

At present, the use of indicators to measure the impact of fishing in marine ecosystems is still in its relative infancy. There is much work still to be done to establish reference limits before indicators can be incorporated into an EBFM framework. In the interim, it is important that indicator studies are designed with a clear objective, that the objective is translated into a testable hypothesis based on sound theoretical knowledge and that the statistical test applied to the hypothesis has a high degree of statistical power. The following section outlines a set of basic requirements for using candidate indicators to model the impact of fishing.

5.1   Define the objective

 The lack of clearly identified objectives is a common criticism of indicator-based management (Niemi and McDonald 2004, Yoccoz *et al.* 2002). In this application, its should be established how best to define the 'success of the CFP'. Much of the comparison of indicators was confused here by the inclusion of parallel target variables such as 'ecosystem status', 'ecosystem health', 'biodiversity' and 'fishery performance'.

A measurable definition of a variable of interest is desirable because it allows indicators to be selected, relationships to be tested and meaningful trigger points to be established. Where possible, simulations and experimental testing are preferable because they can be used to establish a functional relationship between a variable of interest and a proposed indicator.

5.2 Development of a testable hypothesis

Once the objective is identified, it is necessary to develop testable hypotheses for evaluating the relative performance of an ecosystem approach to fisheries management according to the CFP. Examples of testable hypotheses are presented below:-

5.1.1   By-catch and mortality of indicator species

The response of 'vulnerable species' (*eg.* elasmobranches, seabirds, rays and marine mammals) to fishing is often cited as a means of assessing the ecosystem impact of fisheries. By-catches of the harbour porpoise (*Phocoena phocoena*) in the North Sea and Baltic Sea and the thornback ray (*Raja clavata*) in the North Sea have the potential to be used as ecological reference points ICES (2005). The relationship between fishing activity and the abundance of these two species is well-defined, there is a long time series of statistically valid information available and both species are still sufficiently abundant to provide a continuing source of information in the future.

However, not all 'vulnerable' species are suitable for use as potential indicators. It is common to identify vulnerable species on the basis of their response to fishing pressure and such an *a posteriori* approach can only result in a self-fulfilling circular argument. It is more useful to identify potential indicator species *a priori* on the basis that, according to a sound thereotical framework, these species will demonstrate a clear, directional response to changes in fishing impacts (ICES 2005). In addition, there are significant problems with using single species to indicate the status of larger groups of organisms (Roberg and Angelstam 2004). Multispecies approaches have not avoided criticism however. Lawler *et al.* (2003) examined the ability of groups of taxa (*eg.* fish, birds and mammals) to protect wider taxa belonging to the same habitat type. In their investigation, they could find no example of an 'umbrella' taxa in the

entirety of the US mid-Atlantic region. Such studies warn against the use of ecologically sensitive rare species that are often difficult and expensive to sample.

5.1.2 *Species or Population Characteristics*

The response of a species to fishing pressure is determined by their life history characteristics *ie*. whether they are K or r-strategists (ICES 2005). Species with K-selected traits (*ie*. long lived, large bodied, late maturing species) should theoretically respond to increased fishing pressure by demonstrating, for example, a decline in growth rate, maximum body length and both age and length at maturity. The species data can be also aggregated and used to predict the response of the fish assemblage at the community level. In this case, the inclusion of r-strategists alters the theoretical response and characteristics such as growth rate are predicted to increase, while average body length/weight and age and length at maturity should decrease. Trenkel & Rochet (2003) tested many candidate indicators and found that the best performer over a short term in terms of precision and power was mean catch length.

5.2 Statistical tests

The results of the modelling workshop indicate that, at least at present, there are very few indicators that can be analysed in statistical tests that result in a sufficient level of statistical power.

Whatever the statistical test selected (for example, testing for a significant change in reference direction of a linear model, ANOVA for testing for significant variation between groups or ARIMA models for time series analysis) it is essential that the test has a high degree of statistical power. An increase in power can usually be achieved by

- The consideration of the relative costs of Type I and Type II errors and the choice of an appropriate alpha (α).
- The use of 1-tailed rather than 2-tailed tests when there is an appropriate reference direction.
- The analysis of a lengthy time series (usually ≥ 10 years).
- Decreasing the variance in the data (by designing an appropriate sampling survey or considering spatial and temporal factors in the initial phase of data collection).
- Time series modelling approaches to estimate trends in sets of analogous indicators with decision rules based on the modelled trend rather than the observed values.

11

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