PERSPECTIVE / PERSPECTIVE

Current usage of fisheries indicators and reference points, and their potential application to management of fisheries for marine invertebrates[1]

J.F. Caddy

Abstract: The use of indicators in management of invertebrate resources is placed in the context provided by more extensive applications in finfish fisheries. Indicators proposed for the Convention on International Trade in Endangered Species based on extent-of-decline and trend analysis are appropriate should full assessments be unavailable. Measuring reproductive performance frequently builds on egg-per-recruit considerations, given that age structure and stock–recruit relationships are rarely available. Reference points derived from models are compared with direct use of data series, and a broad-brush approach providing a redundancy of indicators is recommended. Indicators may measure productivity as well as biomass and exploitation rate, but ecosystem, spatial, habitat, environmental characteristics, and socio-economic considerations also require monitoring. There is a need to integrate multiple indicators and limit reference points into harvest rules and other decisional infrastructures. The various driving force – pressure – state – impact – response classifications of indicators in use for environmental assessment are now being proposed for marine resources and offer one context for combining multiple indicators. Another is provided by the traffic light approach already used for invertebrate fisheries. The use of indicators and reference points in stock rebuilding is described.

Résumé : L'utilisation d'indicateurs pour la gestion des ressources d'invertébrés est placée dans les contexte des applications plus élaborées qu'on en fait dans les pêches des poissons à nageoires. Les indicateurs proposés par le Convention sur le commerce international des espèces de flore ou de faune sauvage menacées d'extinction basés sur l'importance du déclin et l'analyse des tendances sont adéquats lorsqu'il n'est pas possible de faire une évaluation complète. La mesure de la performance reproductive se base souvent sur le nombre d'oeufs par recrue, puisque la structure en âge et les relations stock–recrues sont rarement disponibles. L'utilisation de points de référence dérivés de modèles est comparée à l'utilisation directe des séries de données et il est recommandé d'utiliser une approche à large éventail qui fournit une redondance d'indicateurs. Les indicateurs peuvent mesurer la productivité ainsi que la biomasse et le taux d'exploitation, mais il faut aussi suivre les caractéristiques de l'écosystème, de l'espace, de l'habitat et de l'environnement et tenir compte des conditions socio-économiques. Il est nécessaire d'intégrer plusieurs indicateurs et points de référence limites dans les règlements de pêche et les autres infrastructures de décision. On propose actuellement diverses classifications de type force directrice – pression – état – impact – réponse des indicateurs utilisés dans l'aménagement de l'environnement pour la gestion des ressources marines et celles-ci offrent un contexte pour la combinaison d'indicateurs multiples. Une autre méthode est fournie par l'approche de type « feux de circulation » déjà employée dans les pêches d'invertébrés. L'utilisation des indicateurs et des points de référence pour la reconstruction des stocks est décrite.

[Traduit par la Rédaction]

Introduction

The collection of data series that support fish and invertebrate stock assessment is a practice that has begun to be formalized in recent years, with the concept of the fisheries indicator coming into prominence. The origin of this change in emphasis can perhaps be traced to the 1980s and 1990s when there was a paradigm shift from a focus on hitting target reference points (TRPs), the specific values of indicators for catch, biomass, and fishing mortality regarded as opti-

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mal, to a precautionary approach that aims to avoid undesirable conditions whose onset is measured by limit reference points (LRPs) (Caddy and Mahon 1995; International Council for the Exploration of the Sea (ICES) 1997; Serchuk et al. 1997), which progressively are measured in other variables, notably reproductive success. The use of indicators and LRPs in management procedures was highlighted by the United Nations Fish Stock Agreement and in parallel by the Food and Agriculture Organisation (FAO) Code of Conduct for Responsible Fisheries. Both regulatory instruments focussed attention on the use of LRPs as guides for management to be incorporated into harvest control rules or management procedures. This new approach reflects a growing appreciation of the low precision possible in determining resource status and hence the risks of overshooting targets. A more explicit role in management decision-making is now given to historical data series in the form of "sustainability indicators" (see FAO 1999; Anonymous 2000). These form elements of the precautionary approach, but as suggested in Department of Fisheries and Oceans (2000), other components incorporated into a fisheries "law" are essential, such that objectives are set, strategies to achieve them are planned and implemented, unacceptable outcomes are defined as LRPs, uncertainty is taken into account, system performance is monitored by indicators, and preagreement is sought from stakeholders on decision rules when limits are approached.

The actual and potential use of fishery indicators and associated reference points (RPs) in harvest control rules for invertebrates is the issue considered here. The "universe" of phenomena encompasses short-lived, motile species such as penaeid shrimps and cephalopods, longer-lived motile decapod crustaceans, and largely sedentary species (e.g., echinoderms, molluscs), some of which (e.g., precious corals) are very long-lived. Attempting to generalize is of course optimistic, considering the broad range of resources, fisheries, and regulatory frameworks that apply, but there are also common elements to the management problem that are also shared to a considerable extent with finfish resources. Among the dangers of extrapolating too freely from one type of resource to another, Seijo and Caddy (2000) and Gilbert et al. (2000) noted that fishery models generating RPs reflect value judgements in their basic axioms. Therefore, decisions on RPs and control laws derived from models whose axiomatic basis does not reflect the fishery context or life history in question can lead to inappropriate conclusions. This raises an alternative possibility of using appropriate indicator values as empirical RPs directly in control laws, and experience on this has been reported from the Southern Hemisphere (e.g., Gilbert. et al. 2000; Breen et al. 2002). In particular, routine application of dynamic pool models to marine invertebrates ignores factors of importance such as the spatial distributions of stock and fishing effort, a key issue for sedentary organisms. Implicit assumptions of stationarity in deriving and applying constant values for RPs may be especially misleading (see Jacobsen 1993) when only a few indicators are used for monitoring. This review therefore advocates use of a broad range of fisheries indicators and RPs to reflect life histories and fishery characteristics, ideally within a transparent fisheries harvest law understood and agreed to by managers and stakeholders.

The use of RPs in managing invertebrate fisheries has a short history so that reference to the more extensive finfish and environmental literature (e.g., Rice 2003) is inevitable. Progress so far with large-scale testing of management procedures on invertebrate stocks is limited (however, see Castilla and Defeo 2001); hence, experience with indicators, RPs, and harvest laws for other resources should be accepted only after careful consideration of the local and species context.

The process of defining indicators and RPs

A diversity of definitions

Indicators measuring events in an ecosystem have their origins in environmental assessment, and I discuss this usage first. Alfsen and Serbo (1993) defined an indicator as "a number that refers to the state, response or the development of important aspects of the environment ... is meant to give information in excess of what is directly measured or observed". Lenz et al. (2000) distinguished between an indicator and an index: "the latter is often constructed from several indicators, weighed together to describe ... the broader state of the environment". In a "Consultation on sustainable indicators for capture fisheries", Garcia and Staples (2000) defined a fishery indicator as "a variable, a pointer, an index related to a criterion. Its fluctuations reveal the variations in those key attributes of sustainability in the ecosystem, the fishery resource or the sector and social and economic well-being. The position and trend of the indicator in relation to the RPs or values indicate the present state and dynamics of the system".

So far in finfish harvest rules, indicators have tended to focus on monitoring fishing mortality, (spawning) biomass, and recruitment. For many invertebrates, the environmental linkage is often more predictive; for example, Barrett and Gillespie (1973) used rainfall, river discharge, and salinity as useful predictors of penaeid shrimp recruitment. Typically, very little of the wide variety of the fishery and environmental data collected routinely is used in deciding on fisheries management measures. Here, the problem is one of how to decide between first- and second-order effects on the managed population or fishery (Fig. 1) and how to organize multiple indicators so that their message is properly integrated and evident to nontechnical stakeholders and managers.

RPs should be regarded as critical values of indicators and may be derived from analysis, from observation, by expert judgement, or by comparison with data from earlier periods in the fishery when productivity was higher and sustained and also, but not exclusively, from population models. The following, based on Halliday et al. (2001), provides useful terminology relevant to the definition and use of indicators. (*i*) Elements of the fishery can be conceptualized as having "attributes" such as biomass, growth rate, natural mortality, accuracy of landing statistics, observance of mesh regulations, etc. (*ii*) A specific method of measuring time series of an attribute results in an "indicator". (*iii*) Groups of indicators that measure related processes are referred to as "characteristics". (*iv*) "Reference points" are values of indicators defined on some technical basis, which are believed to represent important changes in the fishery system. (*v*) Individual indicators may be assigned specific "weightings" and

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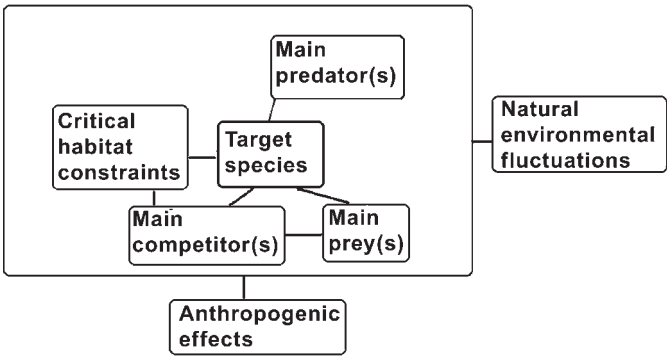
Fig. 1. Some first-order biological interactions directly affecting the managed population that could usefully be monitored in a multispecies fishery and (outside the box) several second-order factors affecting productivity.



"scaled" to normalize them. (*vi*) Individual indicators may be bundled into an "index", which represents the best measure of how a specific characteristic of the fishery is changing with time.

Whether generated from models or from experience, Gilbert et al. (2000) commented that "all the threshold RPs and conservative TRPs in the literature are likewise arbitrary". This is taken to mean that model-based RPs reflect expert judgement at two levels: (*i*) that the model is appropriate to the situation and species applied to and (*ii*) that the indicator value chosen as an RP has some relevance to events in the population, past, present, or future, as judged by experts. At some stage, then, a precautionary judgement must be made that when an indicator approaches an agreed LRP, this corresponds to an unacceptable risk of some negative event occurring. Such an acceptable level of risk must be decided on when reacting to critical indicator values (Prager et al. 2003).

The (D)PS(I)R approaches

Contemporary environmental concerns tend to classify indicators into functional categories, for example, measuring pressure, state, impact, and response (PSIR) (e.g., Malkina-Pykh 2000). Alternatively (FAO 1999; Garcia and Staples 2000; Gilbert et al. 2000), the categories may be reduced to a pressure–state–response (PSR) classification. A broader whole-system approach (see Zenetos et al. 2002) also incorporates economic or environmental "driving forces" into a holistic DPSIR approach aimed at also monitoring socioeconomic, marketing, and regulatory developments as opposed to just the biological components of the fishery system. There is little evidence that such a multiindicator approach to monitoring has yet been integrated into a fisheries management framework, but such a framework is proposed here (Fig. 2). As seen from the title of this section, the terminology is still fluid and practical applications in fisheries are limited. Although "state" indicators could be extended to include environmental, climatic, and pollution effects and multispecies or ecosystem issues, the single-species approach to formulating indicators, based strictly on the target species abundance and its degree of exploitation, still drives most fisheries management approaches.

A statistical framework for indicators and RPs

Managers and stakeholders need to be aware of, and express opinions on, the level of acceptable risk that a management measure will entail (e.g., see Francis 1993; Smith et al. 1993; Shotton and Francis 1997). Ensuring that there is only a low probability of LRPs being infringed requires a probabilistic framework, and Prager et al. (2003) focussed attention not only on errors in indicator values, but also on explicit decisions that need to be made by management on the acceptable level of risk that a LRP will be exceeded.

Normalized indicator values?

Prager (1994) noted the advantages of normalizing time series, or expressing indicator values in dimensionless form as a ratio of current to "optimal" values. Referring the current value of a population variable to some extreme value predicted by a population model (such as the ratio B_t/B_0 frequently used in production model based approaches) is one of a series of fisheries indicators proposed by Gilbert et al. (2000), but M.H. Prager (National Oceanic and Atmospheric Administration, National Marine Fisheries Service, 101 Pivers Island Road, Beaufort, NC 28516, USA, personal communication) suggests that normalizing to extremes of the range of values may incorporate errors into the indicator series, given that B_0 cannot be estimated directly for many exploited species. Using dimensionless or "normalized" indicators is nonetheless relevant, as fishery management is usually a relative procedure: managers rarely have access to the absolute values of fishing mortality rate or biomass in the current or even the previous management season. Thus, adjustments to the target F (or quota) for the next fishing period are often made relative to the indicator values (and quotas) that applied the previous season, and not from a completely new analysis.

If yield is an indicator, this can be "normalized" by comparison with the optimal value from a population model, e.g., maximum sustainable yield (MSY), or an RP representing earlier predepletion yield values. Examples from the literature include the "maximum average yield", which is the long-term average of past harvest levels, or the "maximum constant yield", variously defined as $0.2F_{0.1}B_0$, $0.5F_{0.1}B_U$; 2/3MSY, or cY_U by Annala (1993), in which B_U and Y_U are the "historic" values for biomass and yield prior to overexploitation and c is a "natural variability factor" that ranges between 0.6 and 1.0. Prior fitting of a Bayesian or stochastic model to catch data and repeated runs of the model may help determine how frequently these empirical RPs will be triggered in population management (e.g., Starr et al. 1997).

Prager et al. (2003) formulated indicators and RPs as ratios, which express the current fishing mortality F_{NOW} in relation to an indicator value that corresponds to optima or maxima (such as the MSY, F_{MSY}, or some other TRP). This removes the need to specify q, the catchability coefficient, which is difficult to estimate, and the problem of scaling is also reduced. This may also be appropriate for those indicators (e.g., fishing effort, nutrient runoff, etc.) that do not have a monotonically increasing effect on yield: this increases initially with the effort indicator but declines at high indicator values. Multivariate analysis of multiple time series to determine their functional relationship with fishing yield will in fact be required at some stage in the development of

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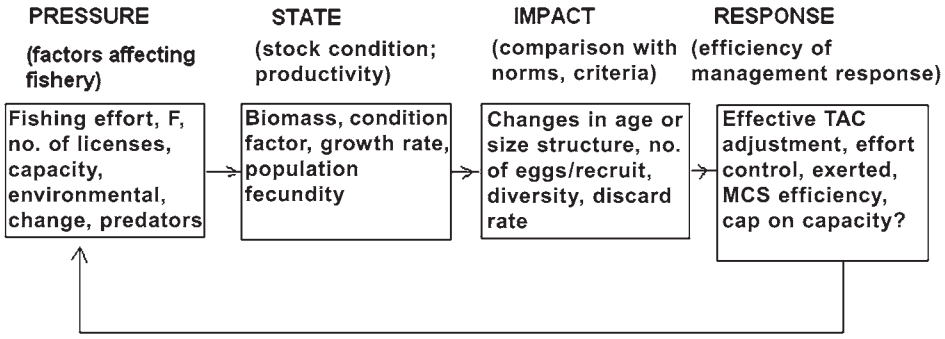
Fig. 2. One possible pressure, state, impact, and response (PSIR) framework for fisheries research and management envisaged in the context of an annual research and management cycle. TAC, total allowable catch; MCS, monitoring, control, and surveillance; *F*, fishing mortality.



a set of indicators. An example of a management procedure using relative indicator values is the Magnusson–Stefansson procedure (Bell and Stefansson 1998) in which next year's quota is seen as a function of this year's yield and the biomass change δB since the previous year.

Errors in interpretation of indicators caused by the incorrect underlying model

Jones et al. (1996) noted that a clearly stated set of objectives is a prerequisite for formulating indicators, especially of ecological impacts: "The crux of habitat management is allowing or prescribing habitat changes that have desirable ecological effects and preventing or mitigating those changes that have undesirable effects" ... "Once these indicators have been determined ... design and implement a procedure for assessing the current status of the indicators, a benchmark against which future change can be contrasted". In addition to errors and biases in measuring variables contributing to the indicator, the choice and interpretation of indicators may be subject to serious errors of interpretation in that the indicator may not be indicate solely what was intended when the "characteristic" in question was defined. Some examples are provided here.

Example 1. Decreases in mean size

Monitoring mean individual size in the population as an indicator of past exploitation rate (which declines as older individuals are fished out by size-selective gear) may be confounded by exceptionally good recruitment. This adds to the argument for juvenile settlement and prerecruit indicators for species such as lobsters (Addison 1997).

Example 2

Mean trophic level was proposed as an index for measuring ecosystem effects of harvesting, namely "fishing down the food web" (Pauly et al. 2001). However, mean size, mean trophic level, and other outputs of food web models emerge as ambiguous measures of exploitation (Rochet and Trenkel 2003) compared with indicators of overall mortality rate or biomass and may decline if high productivity "inflates" the base of the food web as seems to have occurred in the Mediterranean (de Leiva Moreno et al. 2000). It would be more efficient (and less costly in sampling) to use survey data to establish indicators of changes in relative abundance of those predators feeding on the invertebrate species in question and establish a separate indicator monitoring recruitment or spatfall.

Example 3. Thinning

Thinning, a term used for populations of sedentary organisms (see Frechette and Lefaivre 1990), describes the process of actively culling small organisms in dense patches to increase the growth rates of survivors (the harvest of "cherry stone" clams after 1–2 years, leaving the survivors to grow more rapidly to "steamers", a policy giving two economic returns from the cohort). Disease (e.g., Klinck et al. 2001) and starvation can also lead to "self-thinning", which reduces density by intraspecific competition within dense "patches" without human intervention. Density and growth rate of sedentary molluscs are generally inversely related (see reviews in Caddy 1989), whereas natural mortality rate resulting from predator aggregation increases with density. From this perspective, fisheries models of the Beverton and Holt type, which assume that M is constant and independent of age and density, would give incorrect results. Monitoring density in these circumstances may provide a useful proxy indicator for growth and natural mortality rates.

Example 4

Bakun (1998) notes that some spawner–recruit relationships (SRRs) may be "artifacts" of alternating regimes of high and low productivity, with high and low recruitment occurring in different regimes for the same size of parental population. Hence, RPs based on the entire SRR data set may be misleading if stationarity in production is assumed, and recruitment levels should not be extrapolated from favourable to unfavourable regimes for a given stock size (Hilden 1988; Caddy and Agnew 2004). This disguised form of equilibrium thinking is fairly widespread in the literature and has been commented on unfavourably by Gilbert et al. (2000) and Seijo and Caddy (2000). Hilborn (2002) has also cautioned against an undue focus on technical RPs outside their management context.

Example 5

Most groundfish fisheries start after age 2 when predation mortality has largely occurred, but many invertebrate fisheries harvest immature animals for which natural mortality is higher than for adults. Thus, assuming constant natural mortality for juveniles in analytical models or virtual population analyses introduces a bias (see Stoner and Glazer (1998) for

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an example of a declining M vector for *Strombus gigas* and Caddy (1996) for a cephalopod example). When yield models incorporate a vector of M values declining to a constant level at around maturity, the advantages of minimum size regulations are reduced, but the reproductive value of older spawners is increased (see Caddy and Seijo (2003) for a hake fishery, but the principle also applies to long-lived invertebrates).

Categories of indicators and RPs

Different approaches to formulation of indicators and RPs

LRPs and TRPs are critical values of fishery indicators for population variables such as fishing mortality rate and biomass and a series of other biological, environmental, socioeconomic, and regulatory indicators. The following are definitions of common types of RPs following Caddy and Mahon (1995). TRP: "A Target Reference Point indicating a state of fishing and (or) resource which is considered to be desirable, and at which management action, whether during development or stock rebuilding, should be aimed". LRP: "A Limit Reference Point indicating a state of a fishery and (or) a resource which is considered to be undesirable and which management action should avoid".

The notation and use of RPs have since grown more complex, and two types of RP can be distinguished. A generic type of RP distinguishes the intention of the RP and specifies its role in a harvest law. The suffixes of generic RPs such as F_{buf} or B_{lim} (the fishing mortality rate seen as a buffer against overfishing and the lowest safe population biomass, respectively) are examples of this type. More specific or technical RPs such as Z_{1955}, F_{MSY}, $F_{0.1}$, etc., are derived from analysis of (earlier years) data or are outputs from models. Different definitions of generic RPs (e.g., F_{buf} and F_{pa}) have come into use in the North Atlantic Fisheries Organization (NAFO) (Serchuk et al. 1997) and in the ICES (e.g., Hilden 1993), with the ICES definitions showing perhaps a wider diffusion.

Indicators and RPs from trend analysis

A recent FAO technical consultation examined formal extinction criteria for inclusion of endangered aquatic species under the Convention on International Trade in Endangered Species (CITES) into appendix I (species threatened by extinction for which trade is forbidden) or into appendix II (species at risk of extinction unless trade is strictly regulated). Apart from existing CITES extinction risk criteria, the FAO background paper to the consultation (FAO 2001) suggested that indicators related to decline are the most relevant for marine fisheries resources. Three were proposed: (*i*) "recent-rate-of-decline" (the trend over the last 10 years is suggested; also see Musick 1999), (*ii*) "historical rate-of-decline" (the long-term trend over the time series), and, most importantly, (*iii*) "historical-extent-of-decline", which expresses current abundance as a fraction of some "historical baseline", which could be B_{MSY} (the biomass level shown to be capable of safely providing MSY as a sustainable yield), B_0 (the virgin biomass), or, as suggested in FAO (2001), the average of the four largest spawning biomasses in the historical time series.

Taxonomic position was considered by FAO (2001) as less important than life history characteristics in determining risk of extinction, and such characteristics seem well adapted to monitoring poorly studied species for which landing series are the main source of information. A historical rate of decline to 5–20% of the reference baseline, depending on species productivity, was recommended as a general guideline for listing in CITES appendix I, with 5–10% of the unexploited baseline values seen as a safety limit for species with high productivity, 10–15% for species with moderate productivity, and 15–20% for low-productivity species.

For potentially endangered species, FAO (2001) suggested projecting current rates of decline forward over a 10-year period to see if the result meets the above appendix I guideline for historical-extent-of-decline and concluded that historical-extents-of-decline not falling below 50% of the unexploited baseline should rarely be of concern. Given the wide variety of invertebrate life histories, a range of case-specific considerations could modify an overall evaluation based on rate- and extent-of-decline factors. These can be split into vulnerability factors (–) and mitigating factors (+), representing increasing or decreasing concern for the population (Table 1). Such a checklist could also be useful in deciding on indicator series and priorities for management action, as judgements on the need for immediate action will depend on specific life history vulnerabilities or the geographic and (or) socioeconomic situation. An alternative approach to classifying ecological effects of fishing on estuarine invertebrates (Blaber et al. 2000) suggests that indicators fall into the following categories: target and non-target organisms; nursery functions; trophic effects; habitat change and reduced water quality; human environment; and potential for local extinctions.

Other criteria prioritizing needs for conservation by species are given in Nielsen and Kenchington (2001). Although it would be desirable to base indicators on quantitative data, as noted in Seijo and Caddy (2000), an assessment of fishery status in a data-poor situation may use ranked or semi-quantitative criteria such as the scorings from questionnaires completed by experts (e.g., Caddy 1999*c*), comparable with ISO 2000 procedures for environmental assessment.

An example of trend analysis of invertebrate landings

Catch trend indicators for some 150 Mediterranean fish and invertebrates in a complex multispecies fishery were used by Fiorentini et al. (1997) to classify catch trends for which formal assessments were not usually available. These yielded useful indicators for management from catch trends over the short and long term, even without population analysis. As a first step, catch time series from the FAO database FISHSTAT-PC were sorted using a logical sequence of linear and polynomial testing and fitting procedures described in Fiorentini et al. (1997). Each 1950–1995 time series was allocated to one of 10 "shape categories" (Fig. 3) corresponding to segments of a sine curve. This classified for convenience a large number of time series into several long-term trend types, ignoring shorter-term periodicities and year-to-year fluctuations, while a linear trend was also calculated for the last 5 years of each time series (Fig. 4).

What emerged is that a small proportion of Mediterranean finfish and invertebrates fisheries show stable landings.

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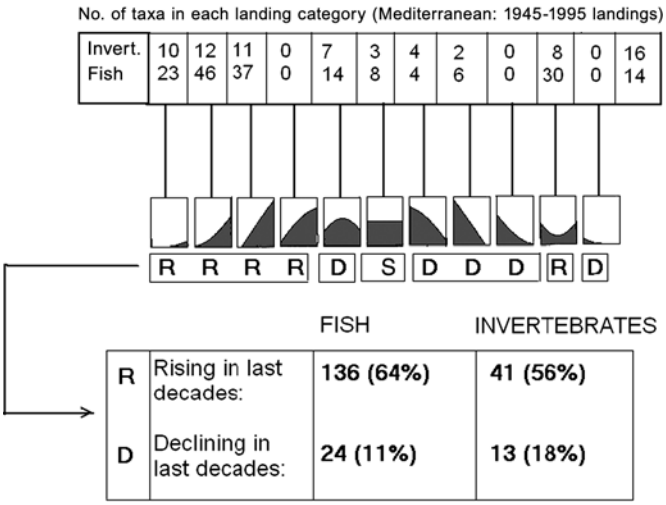
Table 1. Some supplementary factors to monitor that could change criteria based on trend analysis showing the direction of influence of the factor on population vulnerability (modified from FAO 2001).

Description of factor	Sign	Description of factor	Sign
Selectivity of removal (by species or size)	+	Low dispersal of gametes	–
Stock occurs in split jurisdictions	–	Specialized diet, habitat, or niche needs	–
Bycatch in fisheries for other populations	–	Aggregating behaviour or schooling	–
Vulnerable to capture by various gears	–	Endemism	–
Different vulnerability by sex (ratio)	–	Low genetic diversity	–
Source populations very vulnerable	–	Bottleneck in a life history stage	–
High vulnerability at a life history stage	–	Degradation or impacts on habitat	–
Nursery exposed to anthropogenic effects	–	Competing species increasing in size	–
Existence of spawning refugia	+	Predators heavily fished	+
High density critical to reproduction	–	High variance in recruitment	–
Hermaphroditism	–	Unfavourable environmental changes	–
Low fecundity	–	Stock near limits of latitudinal range	–

Fig. 3. Categorization of long-term landing trends by comparison with 11 overall trend configurations derived by dissecting a sine function into components (from Fiorentini et al. 1997). Numbers of species in each category are given for Mediterranean fish (Fish) and invertebrate (Invert.) species separately. The last box (16 and 14) corresponds to an intermittent category with gaps in the time series.



Many species followed long-term upward trends until quite recently despite intensive fishing pressure. This rising trend since 1945 appears not just because of recent fishery developments: the Mediterranean has a long history of diverse fisheries including industrial-scale operations. Landing trends seem to reflect ecological changes underway in a semi-enclosed sea, not all as a result of overexploitation. An increase in productivity of the ecosystem seems the most likely explanation for continued upward trends over the last few decades of high exploitation rate (Caddy 2000). Increased nutrient inputs from densely populated catchment areas to this largely enclosed and formerly oligotrophic sea are documented (de Leiva Moreno et al. 2000). Steadily rising landings of *Venus gallina*, *Pecten jacobaeus*, *Nephrops norvegicus*, and *Squilla mantis*, species typical of fine sediments, of estuarine species (*Carcinus aestuarii*), and of filter-feeding species used for aquaculture such as mussels and oysters suggest that these landing series are indicators of eutrophication. Sponges, red corals, and palinurids, typical of clean oceanic environments, have declined. Overexploitation has become more accentuated in recent years in the Mediterranean, as suggested by the short-term downward trends shown by nine of 27 shellfish species with long-term rising trends and eight of 12 species with long-term declining trends. Of particular concern should be the coincidence of long-term and recent downward trends.

This categorization suggests that a bottom-up effect applies for Mediterranean fauna, namely, an increase in nutrient inputs to a once nutrient-limited inland sea with a very low flushing rate. Invertebrate landing series as a group do not show a completely different range of dynamics from finfish, which presumably reflects similar environmental and fishery impacts and management requirements, although species characteristic of different biotypes can show wide divergences. Evidently, landing trends may be useful indicators of ecological change in specific biotopes.

Monitoring spawning stock biomass and using SRRs or percent spawner-per-recruit models?

Indicator series derived from retrospective analysis and annual biomass surveys are often used to fit SRRs for well-studied Northern Hemisphere fish stocks, but these data sets are rarely available for invertebrates (Caddy 1986*b*). Recent metaanalyses for finfish (e.g., Myers et al. 1994) show that it is no longer safe to assume that spawning populations can be reduced by more than 60–70%, and even these figures may be dangerously optimistic (Caddy and Agnew 2004). Walters and Kitchell (2001) suggested that finfish stock abundance should not fall below 50% unfished spawning biomass, and similar conclusions were reached for abalone by Shepherd and Baker (1998) and for some crustacean resources by Orensanz et al. (1978). For spatially structured populations of sedentary species, SRRs may depend more on local densities than on absolute stock size (Hancock 1973; Orensanz et al. 1990; Orensanz and Jamieson 1998). It has even been argued that a deterministic SRR is not applicable to invertebrate stocks with high fecundity and high density-dependent mortality during their early life history. More recent finfish practice has been to consider the degree of compensation shown by the curvature of the SRR (Iles and Beverton 2000),

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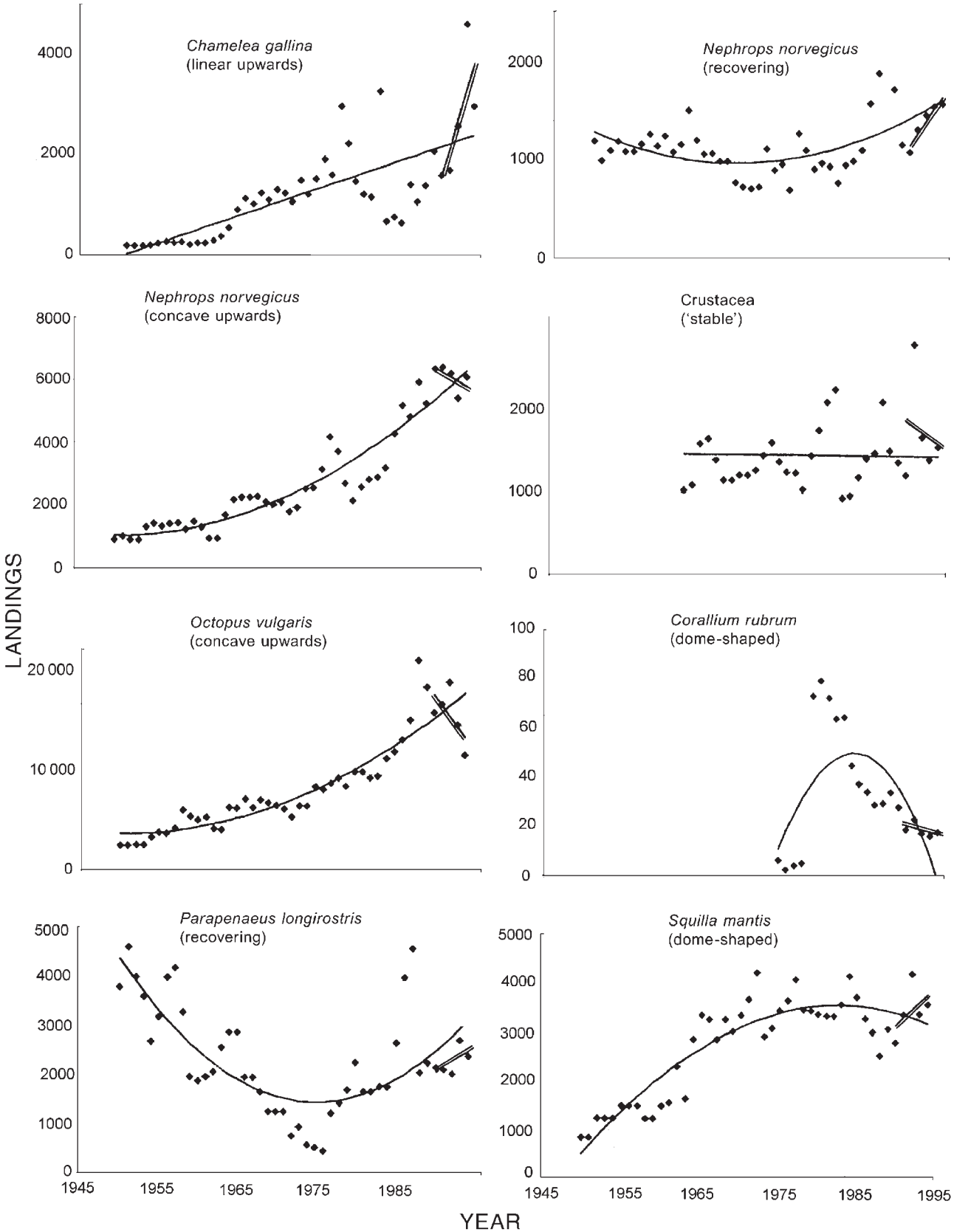
Fig. 4. Some long-term (single line) and short-term (double line) 5-year recent trends for Mediterranean invertebrate landings classified according to the criteria in Fig. 3. "Crustacea" refers to catches not identified to species. (Modified from Fiorentini et al. 1997.)



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and at least for abalone (Shepherd et al. 2001), SRRs are flat and the possibility of depensation (see Liermann and Hilborn 2001) is suggested by the failure of depleted abalone grounds to recover after 15–20 years.

Percent spawner-per-recruit (%SPR) theory (Gabriel et al. 1989) requires that successive generations produce sufficient "spawning units per recruit" (expressed in biomass or eggs) over their life span to replace themselves. Obviously, this is not easy to estimate, nor is it an absolute criterion. If regime shifts reduce productivity, recruitment may decline, and hence, some experts prefer to calculate the actual number of eggs produced by the population. Deriving RPs from %SPR calculations is widely used in finfish management (see ICES 1993; Mace and Sissenwine 1993; Mace 1994). Rates of fishing on abalones that do not reduce spawning potential to below 50% of unfished levels ($F_{50\%}$) were believed appropriate for small stocks and $F_{40\%}$ for large stocks of abalone (Shepherd and Baker 1998). A safe limit of 30% virgin population fecundity was estimated for *Haliotis rubra* by Sanders and Beinssen (1998), below which simulations showed that stock recovery could not be assured. Species such as squids with annual life histories may also require high thresholds (e.g., Agnew et al. 1998). In contrast, conventional wisdom is that high-fecundity spawners such as many sessile molluscs may recover from low thresholds, but even for these, evidence suggests that recruitment is enhanced by healthy source populations. Threshold values of $F_{10\%}$ have been suggested for *Homarus americanus*, whereas the overfishing definition suggested for *Panulirus argus* by the National Marine Fishery Service is triggered when the egg-to-recruit ratio is 5% of the unfished population and recruitment has been falling for 3 consecutive years (Rosenberg et al. 1994). In all cases, calculating an indicator based on relative fecundity of the current to unexploited conditions seems a feasible approach.

A range of "guesses" as to the limiting spawning potential of invertebrates are given (Fig. 5), also incorporating values for pelagic and demersal finfishes for comparison. Like vertebrates, invertebrate species span a wide range of relative fecundities from which populations are believed to be recoverable, but Fig. 4 contradicts the common assumption that invertebrates will always recover from low stock sizes. Whether a scallop population could consistently recover from 2% of the virgin stock size as tabulated by Mace and Sissenwine (1993) probably depends on local density and the degree of aggregation of the remaining spawners as well as on environmental conditions (Haywood and Staples 1993) and not just on total stock size, and the degree of compensation possible after stock depletion may vary significantly (Fig. 5). With few exceptions (e.g., Ennis and Fogarty (1997) for *H. americanus* and Shepherd and Baker (1998) for *Haliotis laevigata*), SRRs have rarely been derived for invertebrates. Effects of environment are often more convincing than the effect of spawning stock size on invertebrate recruitment, and for bivalves, local adult biomass may inversely affect local settlement success (Hancock 1973). The derivation of model-based RPs for invertebrates could follow %SPR or fecundity-per-recruit approaches, but as noted, for sedentary species, local density and location rather than just population size are probably the key variables. Some reservations nonetheless have been expressed concerning the use of %SPR and fecundity-per-recruit measures for lobsters and other long-lived species, as the calculated fecundity of unexploited stocks used for reference in egg-per-recruit calculations requires assumptions as to M values at age, moult frequency, and the mating success for very large lobsters, values that it is rarely possible to verify. It would be more convincing to use indicators comparing fished populations with those in unfished marine protected areas (MPAs). The importance of older lobsters in reproduction has been stressed (Caddy 2002) as for finfish (Conover and Munch 2002), and an indicator estimating mean age or size of mature females could be useful to measure truncation of mature age groups.

Indicators and RPs based on spatial and habitat factors

The term "productive capacity" (Jones et al. 1996) is commonly used to describe "both the carrying capacity of an environment, including the productivity of organisms on which *the species* depends". Indicators reflecting production per habitat area or carrying capacity are especially relevant for invertebrates (e.g., Bolonga and Steneck 1993; Herrenkind et al. 1997) and are critical for sedentary species. Density emerges as a key variable, and Botsford et al. (1993) noted that below 1 female·m^{-2}, successful fertilization of *Strongylocentrotus franciscanus* is limited, and minimum densities for successful fertilization must apply to most sedentary species. Shellfish populations often show contagious distributions such that much of the stock lies within a small proportion of the stock area, which in addition to being the "source" of new recruits is also the most profitably fished component. Indicators of growth rate and mortality should ideally be quantified through time (i.e., under different densities) and in space (by fishing ground) to evaluate variations in density-dependent processes and habitat quality.

The issue of source and sink populations is often referred to for sessile and semisessile invertebrates. For example, the role of source populations for the Georges Bank scallop (*Placopecten magellanicus*) stock was commented on by McGarvey et al. (1992) who found that 82% of egg production came from a small area of the bank and was produced there at a high density. This argument was also invoked for a crustacean, *P. argus*, by Lipcius et al. (1997). The suggestion was made (Caddy 1989) that range extensions during favourable regimes may be extensions of impermanent "sink" fringes. Thus, estimates of range represent expansions and contractions of populations and are useful indicators of productivity (see Caddy and Defeo (2003) for an example on surf clams). It is also suggested that indicator values for density and biomass be collected where possible from source populations.

For invertebrates where metapopulations are the rule (i.e., individual shellfish beds within a fishing area), the concept of a unit stock seems to break down, and even if genetically homogenous, not all segments of the metapopulation are of equivalent value from a population perspective. Mesoscale (local populations) and macroscale or megascale (geographic) patterns are usually present, and some grounds with regular recruitment have a higher probability of sustainable yield and may form "source" populations for adjacent areas (Caddy and Defeo 2003). Especially for sedentary invertebrates, it is important to distinguish between source populations that produce most of the progeny and "sink" areas that

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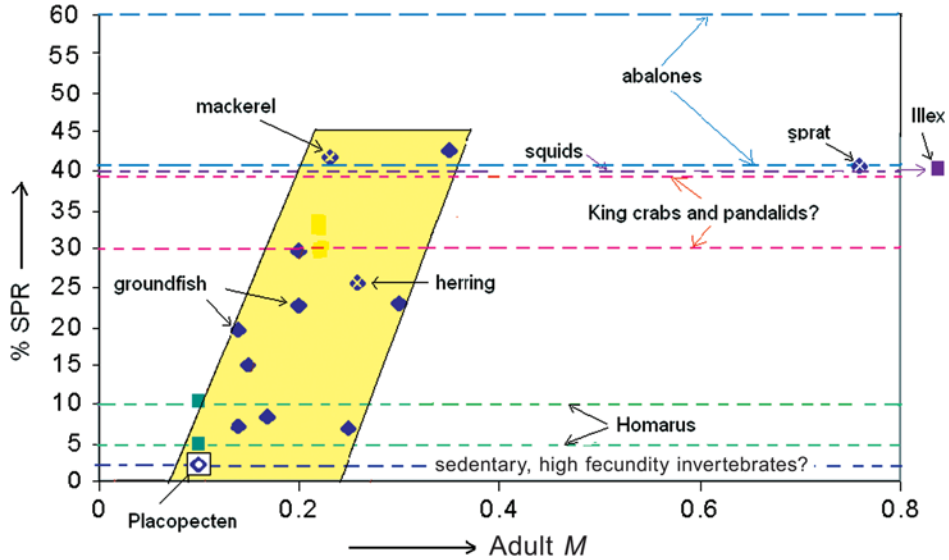
Fig. 5. Postulated ranges of minimum values for percent spawner-per-recruit (%SPR) for fish and invertebrates from which stock recovery is believed possible. Data for a range of finfish %SPRs (most commercial finfish appear to fall within the parallelogram) are from Mace and Sissenwine (1993) and are plotted against natural mortality (*M*) rates mainly from FISHBASE.



in good years may receive "vagrant" juveniles, as local populations may not be self-perpetuating even though they form part of a metapopulation with a common gene pool (see Orensanz and Jamieson (1998) for a discussion of spatial factors in shellfish populations). If these two groups could be distinguished, it would be legitimate to harvest sink areas without major control as long as the "mother lode" of spawners in the "source" area is protected. It may even be a viable strategy to build up spawner density there, as for many bivalves, echinoderms, and abalones, gamete fertilization is distance constrained, or as for abalone, juvenile dispersal may remain local to the parent population (Shepherd et al. 2001). One criterion for recognizing source areas is to regularly map recruitment areas and protect those where a more even distribution of age classes occurs.

For macrocrustaceans, nursery and spawning areas and migration routes need to be distinguished in management, and serial depletion of crustacean populations was documented by Orensanz et al. (1998) and seems to reflect spatial vulnerability of source areas for these species. An important survey-based indicator, then, is the proportion of the stock area containing densities over some limiting value. Appropriate scales of sampling must be used to characterize the large variation in dispersal dynamics over space and time and the between-site variation in settlement dynamics of larvae. Single "snapshot" studies generally lead to erroneous conclusions; hence, the need to construct indicator series from repeated observations. Life histories and the characteristics of habitat or substrate may cause recruitment bottlenecks and density-dependent processes (Wahle and Steneck 1991). If estimating fishing mortality rate is a problem, as for many semisedentary species, a spatial exploitation strategy could be the solution. Should yield-per-recruit suggest an optimum target *F*, it may be possible to approximate this by a rotating harvest scheme (e.g., Caddy and Seijo 1999) (see Fig. 6).

Even for motile invertebrates, indicators monitoring nursery areas such as described for *H. americanus* (Wahle and Steneck 1991), *Panulirus* (Acosta and Butler 1997), egg brooding areas for *Cancer magister* (Scheding et al. 2001), and nesting areas (*Octopus vulgaris*) may require special protection and monitoring. Such areas of critical habitat may be more constraining on the effective level of recruitment than trophic considerations and, in some circumstances, perhaps even than minimum spawning stock size. In fact, once a bottleneck is passed in the life history, food and living space may not be the critical constraints (Fig. 7). This suggests that stock enhancement by increasing egg production may not always be the priority, but protecting or enhancing nursery area may be. Mapping and geographic information

Fig. 6. A rotating harvest scheme in which a nonmigratory stock is divided into subareas with approximately equal proportions of stock in each area. The number of areas is equal to, or a multiple of, the rotation period. Variable harvesting rates (60–90%) in the open area by the end of the season are calculated to result in a given *F*-based target reference point (TRP) = 0.2. Curved lines from top down represent 90% (◇), 80% (■), 70% (▲), and 60% (×) of stock harvested in the open area during a year. (From Caddy and Seijo 1999.)

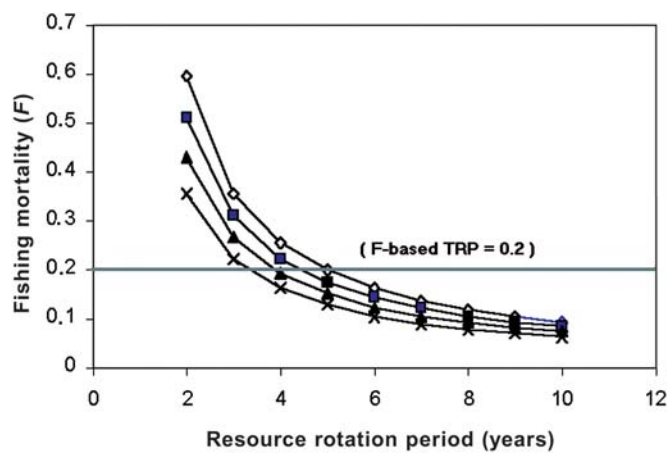


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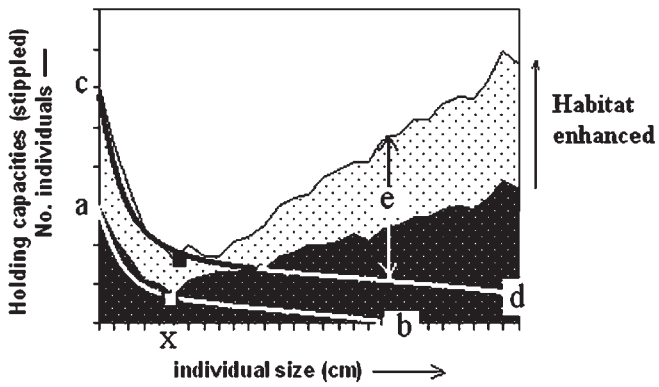
Fig. 7. Illustration that a typical survival curve (*a–b*) will limit survival of adults to below the carrying capacity of the habitat if limited habitat forms a bottleneck at critical juvenile size *X*. Although enhancing habitat holding should improve survival (*a–b* improving to *c–d*), the holding capacity of juvenile habitat is critical, and much suitable adult habitat (*e*) will likely remain unused. Thus, indicators of suitable habitat size could usefully focus on juvenile stages.



systems applied to monitoring critical habitats, their area, and (or) the densities within them are likely to provide fishery indicators with high predictive significance if aggregations of juveniles or spawners occur. The use of overlays to determine degree of interaction between ecosystem components is also potentially a source of useful spatial indicators (e.g., Nishida and Booth 2001).

Measures of year-class regularity

Alternative control laws have been based on an indicator monitoring the proportion of the biomass consisting of older animals. Rosenberg and Brault (1991) suggested such a simple measure of year-class regularity based on the skewness of the age distribution, noting that as exploitation increases, the age composition becomes more skewed, as the population consists of fewer year classes. They defined

$$\text{Skew} = (1/N)\sum[\ (X_i - X_{\text{bar}})/s]^3$$

where N is the number of age groups, X_i for $X_1,..., X_n$ is the relative abundance of the ith age group, X_{bar} is the average relative abundance, and s is the standard deviation of the relative abundance. The skewness relative to a value calculated for the unexploited stock might be a useful criterion for judging the success of recovery strategies, i.e., aiming for a given relative skewness as a criterion for rebuilding. The skewness of local age compositions could also be a spatial criterion for distinguishing "source" areas with regular recruitment from "sinks" with only occasional year classes.

The problem of deciding on a common management framework for metapopulations arises frequently for spatially differentiated invertebrate stocks, and Maritime Canada lobster districts, for example, may or may not correspond to separate stocks, and different local management approaches have evolved. Reflecting the fact that few adult lobsters survive to reproduce, a common metapopulation criterion was proposed by the Fisheries Resource Conservation Council (FRCC), namely to increase egg-per-recruit by a uniform 1% in all lobster management districts while allowing some leeway as to how this might be accomplished locally. Thus, individual areas may have a first tier of management measures representing local stakeholder preference where monitoring requirements can vary, but a common element for the supposed metapopulation may be a fisheries control law requiring an overall increase in egg production to be satisfied.

Productivity indicators and their RPs

Indicators and RPs used for finfish management are usually restricted to catch, biomass, and fishing mortality. This may partly explain the poor performance of harvest rules based on limited data inputs in the face of ecological change and regime shifts. A wider range of variables appears to be required for monitoring and managing invertebrate fisheries, and indicators of fishery productivity seem a key option to consider. FAO (2001) defined productivity as "a complex function of fecundity, growth rates, natural mortality, age at maturity and longevity", and productivity seems a reliable surrogate for population resilience. More productive (short-lived) *r*-selected species have high fecundity and growth rates and a rapid turnover of generations and from low numbers can rapidly take advantage of conditions suitable for re-establishment; *K*-selected species with low productivity tend to spend longer periods at low population size once depleted and run greater risk of extinction from depensatory factors. FAO (2001) considered that "the most important property of species and populations relative to risk of extinction is their resilience and ... this is best reflected by the productivity of the species".

Mohn (2002) remarked that productivity is given insufficient emphasis in standard stock assessment, and in fact, a built-in "equilibrium assumption" often seems to rely on recruitment and surplus production being constant, and noted that productivity can also be expressed in terms of the age-specific rates of growth (g), natural deaths (m), fishing (f), and recruitment (R), so we might write

$$B_{T+1} = B_T \exp(g - m - f) + R_T$$

where year-specific subscripts for growth and natural and fishing mortality rates are omitted for convenience, although rates may change from year to year because of changes in environment, food availability, and predator abundance. This equation is not easily solved but suggests that a "characteristic" measuring stock productivity should incorporate indicators reflecting changes in biomass, recruitment, and fishery yield and also monitor, if possible, changes in natural mortality rate and growth.

Quantifying productivity is difficult, but Table 2, based on FAO (2001) and criteria proposed by Musick (1999), suggest a functional classification into three production categories that could also be adopted for invertebrates. Although the International Union for Conservation of Nature guidelines based on small population size may only apply to sessile, semisessile, and extremely low productivity species, where a bottleneck exists at some point in the life history, critical habitat area for some life history stage can be important (Fig. 6). This would be the case if a species passes its (early) life history in a restricted or fragile habitat, such as estuaries, grass beds, or mangrove swamps, where changes in area and density could be indicators of potential productivity. For

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Table 2. Guidelines proposed by Musick (1999) and FAO (2001) for classifying finfish species by productivity (estimates in parentheses from Musick 1999).

Parameter	Low productivity	Medium productivity	High productivity
M (natural mortality rate)	<0.20	0.20–0.50	>0.50
r (intrinsic rate of increase) (year^{-1})	<0.14 (0.05–0.15)	0.14–0.35 (0.16–0.50)	>0.35 (>0.50)
k (von Bertalanffy coefficient)	<0.15 (0.05–0.15)	0.15–0.33 (0.16–0.30)	>0.33 (>0.30)
t_m (age at maturity) (years)	>8.0 (5–10)	3.3–8.0 (2–4)	<3.3 (<1)
T_{max} (maximum age) (years)	(11–30)	(4–10)	(1–3)
Finfish species	Orange roughy, sharks	Cod, hake, plaice	Sardines, anchovies
Possible invertebrate examples	Red coral, abalone, lobsters, Tridacnidae?	Small crabs, *Nephrops*, some pandalids?	Penaeid shrimps, cephalopods

Note: Invertebrates suggested to be added to each category are shown in the last row.

molluscan shellfish, productivity may also depend on environmental inputs such as phytoplankton abundance, seasonal oxygen minima, turbidity, nutrient inputs, etc., and the criteria proposed by Alfsen and Serbo (1993) are relevant here. The critical importance of density as an indicator affecting growth rate, natural mortality (by predation or disease), and spawning success needs to be considered.

Ecosystem and environmental indicators

The existence of "top-down" effects as suggested by Pauly et al. (2001) raises the question of whether top predator abundance is a useful indicator of predatory impact. Invertebrate fisheries where this is apparently the case are interactions of sea otter and abalone (Tegner 1989), northwest African sparids and *Octopus* sp. (Caddy and Rodhouse 1998), gadoids and *Nephrops* in the Irish Sea (Brander and Bennett 1989), and red grouper and *Octopus* sp. (Arreguin-Sanchez 2000). This raises the questions for what species and in what areas and seasons do critical interactions occur and how can they be monitored with meaningful indicators. In the case of gadoids and Norway lobsters (*N. norvegicus*), Brander and Bennett (1989) presented a multispecies bioeconomic yield model that suggested using predator biomass (or perhaps the predator-to-prey ratio) as an indicator in *Nephrops* fishery management. However, Kruse and Zheng (1999) found no obvious correlation between biomasses of groundfish and king crab decline, and Hanson and Lanteigne (2000) reached similar conclusions for cod and lobsters in the southwest Gulf of St. Lawrence. Nonetheless, the conclusion seems inescapable that in some cases, predatory finfish abundance may be a negative indicator for invertebrate fishery productivity. Collie and Gislason (2001) suggested restricting multispecies considerations to the immediate target species and its main predators and prey and considering one-way flows between them, and such empirical indicators are implemented by Link et al. (2002). Thus, predators can be added to single-species prey assessments without considering the totality of ecosystem interactions. The problem of deciding on suitable indicators is, on the one hand, constrained by the usually limited data series available and, on the other, due to the difficulty of deciding on first- and second-order effects of the indicators believed to be relevant (Fig. 1).

The maximum biological production (MBP) is a TRP for productivity that Die and Caddy (1997) showed to be safer than MSY, and for managing prey species, Collie and Gislason (2001) suggested keeping the total mortality rate (Z) below a threshold such that the total allowable catch in year t (TAC_t) is defined by

$$\mathrm{TAC}_t = Z_{\mathrm{MBP}} B_{\mathrm{av}_t} - \mathrm{PC}_t$$

where Z_{MBP} is the mortality rate at maximum biological production, B_{av_t} is the projected mean prey biomass in year t, and PC_t is the prey biomass consumed by predators.

The key role of environmental conditions in controlling invertebrate production needs to be recognized in decision-making, as yield largely depends on recruiting year-class size, which for invertebrates is often a function of biotic and abiotic factors but also of anthropogenic effects such as nutrient runoff or incidental impacts of dredging on epifauna (e.g., Pitcher et al. 2000). As highly fecund invertebrates are often low in the food web and sensitive to bottom-up effects, indicators measuring plankton productivity, turbidity, oxygen levels, and eutrophication should be useful in predicting the typically large variations in recruitment success that drive these fisheries.

Indicators derived from biological characteristics

Crustacean biological processes are dominated by moulting, which generally decreases in frequency with age. Hence, a decline in the abundance of older animals with fishing pressure leads to a higher frequency of moulting and a higher proportion of animals in a soft-shell condition, which emerges as an indirect index of exploitation rate (Caddy 1986*a*). The change from juvenile to adult morphological characteristics can also be used as a proxy for age composition, e.g., for conch (*S. gigas*) or whelk (*Buccinum undatum*) populations from the proportion of thick-lipped shells in the catch, as thickening occurs at maturity. Thus, it seems feasible, and probably cost-effective, to develop what may be called "quality indicators" such as sex ratio, annual recruitment, proportion of mature fish in the stock, etc., and measure them by special sampling schemes or from surveys. This type of indicator, coming from an independent sampling from catch rate or fishing effort, should not be subject to the confounding effects that plague production models in which a plot of catch-per-unit-effort against fishing effort compares two variables derived, in part, from the same indicator series.

Fu et al. (2001) investigated the role of sex change in population dynamics and management of a protandrous hermaphrodite, *Pandalus borealis*, in the North Pacific. As

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Table 3. Four possible indicators for each of four characteristics of a traffic light monitoring system for hypothetical crustacean species A with predators 1 and 2 and co-occurring species B.

Indicator	Characteristic
1. Mean survey catch per trap	Abundance
2. Area with density >5·m^{-2}	
3. Early-season catch per trap haul	
4. Bycatch species A in trawl fishery for species B	
1. Number of recruits (carapace length <5 cm)	Production
2. Area of recruit density >10·m^{-2}	
3. Mean size of mature females	
4. Condition factor (carapace length = 10–15 cm)	
1. Mean Z_t from survey data	Fishing pressure
2. Fleet days fished per season	
3. Immature individuals (%)	
4. Annual number of trap hauls per area grounds	
1. Abundance (predator1/species A)	Ecosystem/environment
2. Abundance (predator2/species A)	
3. Absolute value |bottom temperature – optimum temperature|	
4. Prey abundance·m^{-2}	

mainly females are exploited, the sex ratio drops with heavy exploitation and impacts reproductive success, and hence, this could be a useful indicator. A related question is raised: what is the optimum age at which to take a cohort of females that is bearing eggs? This has also been a leading question in the Gulf of St. Lawrence lobster fishery. The importance of retaining spawning potential for large fecund crustaceans (for which larger females are less vulnerable to capture) is illustrated by American lobster management in the Maritimes. Some ideas for biologically based RPs for a macrocrustacean are given (Table 3) based on Caddy (2002).

Is there a case for "empirical" RPs?

The examples given earlier illustrate that model-based RPs may be misleading if the model incorporates assumptions at variance with species biology, environmental change (i.e., stability), and demography. Defining RPs using precaution and past experience with the fishery is not excluded in "real-life" management, and for some invertebrates, there are few alternatives to this approach. Conventional fisheries indicators often include fishing intensity (= effort per unit area), with raw data preferably adjusted for fishing power by fleet components. Estimating total catch usually assumes that there is a statistical sampling scheme in place that collates the raw data into an annual figure for each stock unit and that an annual survey programme estimates biomass. Estimates of total catch and fishing effort that depend on sampling the whole fishery or on a census of all fleet operations are particularly liable to bias because of misreporting. Special schemes to sample life history characteristics such as sex ratio, percent soft-shelled animals, etc., may also provide useful and cost-effective indicators.

The use of empirical RPs derived directly from those "historical" values of indicator time series that marked the onset of earlier negative fishery situations is an alternative strategy. Such RPs may also emerge from discussion between experts, managers, and stakeholders. Although less "tidy" scientifically, empirical RPs may represent past unfavourable conditions remembered by stakeholders and thus provide an easily understood incentive to work towards stock reconstruction if this is required.

Although judgement rather than analysis is frowned upon in science, indicators and RPs, strictly speaking, are management tools, and their significance should be understood by stakeholders and interested parties if they are to be effectively used in a management rule. The Law of the Sea and subsequent international legal instruments specify that management action should not be delayed but based on "the best information available" but do not indicate that predigesting fishery indicators through a modelling procedure is mandatory. Rightly or wrongly, judgement on the basis of inadequate data is routinely applied in fisheries management, although it would be greatly improved if formal or informal rules for using information from indicators were incorporated in decision-making. This does not exclude model-based RPs, but some schools of practice use stochastic modelling mainly as a means of testing harvest laws and the (often empirical) RPs that they contain to determine their performance and precautionary nature (e.g., Zheng et al. 1996; Breen et al. 2002) rather than just as a source of RPs.

Use of indicators and RPs in fishery control laws

There are three overall types of control laws driven by stock indicators. (*i*) A constant low total allowable catch (TAC) strategy requires a high escapement and high stock biomass and productivity. Similar strategies are used in New Zealand fisheries, such as the maximum constant yield (Annala 1993) in management of New Zealand rock lobsters. Although constant catch strategies may not be ideal, the difference between total catch and the preset TAC is an essential "response" indicator in quota management. (*ii*) A constant fishing mortality strategy is often considered safer

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for finfish stocks but raises the problem of monitoring fishing power of the fleet having access. This strategy results in a variable spawning stock size and variable catches, especially for stocks with irregular productivity. Sanders and Beinssen (1998) found that the use of TACs to target a fixed exploitation rate is an uncertain method of stock rehabilitation for abalone if recruitment is very variable. Maintaining effort control directly may give more consistent results, but then, increases in fishing power through new technology need to be monitored. Adding a minimum biomass threshold for closure during a low-productivity regime is precautionary so as to avoid overfishing during poor recruitment years even where a cautious (e.g., $F_{0.1}$) strategy is pursued. A ramp function progressively reducing exploitation rate when biomass falls below a precautionary level is the COMFIE approach (ICES 1997). (*iii*) Aiming for fixed escapement to spawning is an option suggested for a prawn fishery (Caputi 1992) and has also been used for short-lived species such as squids (Basson et al. 1996); the Falkland squid fishery (Agnew et al. 1998) targets a fixed escapement to spawning of no less than 40%. This approach requires fishery yield to entirely absorb stock variations from year to year.

Other variants to these three types of control laws are possible, but whatever management regime is in place, a recovery strategy will be called for when stock size falls below an LRP for spawning biomass.

The traffic light approach

The traffic light (TL) system (Caddy 1999*a*, 1999*b*) offers a way of using multiple indicators and their critical RPs for managing populations of invertebrates for which age structure information and SRR-based RPs are unavailable. Initially, a range of indicators measuring quantifiable life history characteristics were envisaged as scored into red or green if they fall on one side or the other of an appropriate LRP. With multiple indicators, the proportion of indicators in the "red" zone might determine the increasing severity of management response. Following intensive consultations (Halliday et al. 2001), the concept was modified to include the use of yellow if indicator values represent an uncertain situation as transitional between a safe or "green" condition of an indicator and "red" indicator values, representing a dangerous condition for the stock. Various approaches were explored for dealing with transitional and boundary situations involving fuzzy logic, ramp functions, etc. The problem that emerges is not just how to define RPs (or in TL terminology, colour boundary values) but, when using multiple indicators, how to weight them before combining them within a "characteristic" for (say) biomass or fishing mortality. No general rules can be offered at this point, but the advantages of working with multiple indicators is evident even if the TL approach is simply providing a diagnostic or index of "ecosystem health", as in impact assessment. Koeller et al. (2000) used this approach for characterising shrimp fisheries, and Halliday et al. (2001) suggested that we refer to this diagnostic tool as a "traffic light stock status index". Combining multiple indicators within a formal management rule is a more complex issue, and the key objective of the TL system is that it should be integrated into a precautionary harvest law, or at least a management information system involving scientists, managers, and stakeholders. One approach proposed by Halliday et al. (2001) is that decision rules should be based on the integrated score of indicator values measuring at least three characteristics: abundance, production, and fishing mortality. Because colour boundaries of individual indicators of a characteristic (e.g., see Table 3) may not be triggered simultaneously, a gradation of response is likely to result, which should provide some redundancy and "smoothing" if the proportion of indicators triggered within a management rule determines the severity of management response (Caddy 1999*a*, 1999*b*).

Other performance measures could be included in a multi-indicator fisheries monitoring system; for example, indicators should ideally monitor key socioeconomic criteria, fleet capacity, and the efficiency of control and surveillance measures (e.g., Fig. 1), with centres of responsibility operating within a fishery management framework monitoring these different indicators as performance measures.

Halliday et al. (2001) suggested that fisheries productivity be given high prominence as it drives fisheries yield and proposed tying this TL diagnostic function to a quota rule. Separate indicator values for productivity, biomass, and fishing mortality could perhaps drive a fuzzy logic decision framework. Some excerpts from a set of quota change rules tied to a TL system might include the following:

IF Production = green AND Fishing mortality = green AND Stock = green THEN TAC increment can be "small positive" BUT:

IF Production = green AND Fishing mortality = yellow AND Stock = yellow THEN TAC increment is "no change" OR:

IF Production = yellow AND stock = red THEN TAC increment must be "large negative".

Changes in the coming year's TAC would thus be determined by changes in three sets of characteristics and their respective indicator values. The resulting quota changes relative to last year's allowable catch would be dictated by precautionary rules: favourable conditions allowing a small increase in quota but unfavourable changes resulting in a larger cut, the opposite tendency to that often produced by conventional management in the absence of capacity control, which tends to result in continual increases in exploitation pressure. An alternative approach suggested informally by the FRCC is to tie the TL diagnostic to a "consideration matrix". This requires scientific advisors to place the current fishery within one of a number of cells in the matrix, each corresponding to a combination of stock condition and productivity regime. The managers must then operate within the constraints for allowable quota change prespecified within each "box". Although the scientific component and multi-indicator approach with these types of strategy appear less precise than when quota management is driven by formal assessments or other population analytical tools, the procedure emphasizes that the most important contribution to the rigor of a management rule comes when decision rules seriously constrain management choices.

A similar system of multiple indicators was the "trouble spot thermostat" proposed by Shepherd et al. (2001) for an

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abalone fishery, which is a good example of using multiple indicators in a harvest law for the data-poor shellfishing grounds making up a metapopulation. This protocol for managing individual fishing grounds of greenlip abalone uses diver survey, and analysis of commercial shell samples allows a response to declines of local populations. Recruitment time series emerged as sensitive indicators, as did the total catch from, and degree of spatial contraction of, individual subpopulations. Catch-per-unit-effort, size frequency, and total mortality (Z) proved to be relatively insensitive indicators of population decline.

Indicators and RPs within stock rebuilding strategies

High-value stocks, as for many invertebrates, are eventually fished down unless an appropriate management framework is in place. Perry et al. (1999) postulated three phases in scientific information gathering in support of management: (*i*) collecting and synthesizing existing information, (*ii*) collecting new information and evaluating alternative management strategies, and (*iii*) monitoring commercial fishing while the chosen management actions are implemented. Less optimistically, given extensive experience with South American shellfisheries, Castilla and Defeo (2001) suggested that development of shellfish fisheries tends to follow six phases: (1) initial exploitation phase, (2) expansive extraction phase, (3) overexploitation phase, (4) closure phase, (5) stabilization of extraction and institutionalization phase, and (6) mature and consolidation phase.

Planning a formal management approach for many fisheries often awaits phase 3 or even phase 6, and initiating a recovery plan in phase 6 requires a lower, sustainable level of exploitation by a limited number of stakeholders with, by now, rights-based approaches such as individual transferable quotas. All phases need indicators to be monitored, but recovery simulations require an approach to the use of RPs that differs from "business as usual" management but must be understandable to stakeholders. The end point for a stock recovery plan is a TRP (e.g., B_{MSY}), expressed in spawning biomass. This should be higher than the LRP infringed in the stock decline; otherwise, the fishery will simply oscillate around this unsatisfactory level once recovery has been achieved (Caddy and Agnew 2004). During stock rebuilding, sequential targets expressed in terms of biomass- and F-based TRPs should dictate a recovery trajectory and attempts should be made by sampling and surveys to determine the probability that this is being maintained. It is now common practice to simulate the fishery as a way of testing recovery strategies before application, but unlike MacCall (2002), most recovery plans assume that the fishery is the key controlling factor, and rarely is specific account taken of regime shifts or ecosystem changes that may contribute to stock collapses; hence, unrealistically short recovery times may be predicted. Five phases can be recognized in a rebuilding plan: (*i*) the (spawning) stock biomass indicators drop below LRPs established previously during routine exploitation, and this initiates the recovery plan (already prenegotiated with stakeholders), (*ii*) a recovery trajectory is defined as a series of (preferably) time-specific TRPs for fishing mortality and biomass (although the recovery trajectory may be accelerated or slowed depending on recruitment), (*iii*) a recovery TRP for biomass is defined, which when reached will terminate the recovery plan, (*iv*) when the biomass indicator is back above the recovery TRP, optimal harvesting may resume at levels applied before collapse, but a more precautionary approach would carry forward aspects of the recovery plan, for example, the use of multiindicator monitoring of productivity, and (*v*) indicators and their RPs should be incorporated into the legislative framework for the fishery.

Apart from slow-growing species, those with low larval dispersal and (or) species with restricted or vulnerable habitats such as the Tridacnidae, pearl oysters, *Pinna* spp., and holothurians, stock rebuilding has not been the major problem for most invertebrates that it has been for marine fishes. Despite this, a significant number of invertebrates and their local stocks may be in danger of extinction (see Wells 1989). Rebuilding approaches have tended to focus on closures (see Castilla and Defeo 2001), although for some invertebrate stocks, stock building seems more a consequence of the collapse of predatory fish populations than of a deliberate building plan (Caddy and Agnew 2004).

Discussion

The transition from dominance by TRPs as implied by the Law of the Sea to precautionary LRPs as recommended following the United Nations Conference on Environment and Development and the 1995 United Nations Fish Stock Agreement has led to a focus, at least in finfish science, on RPs generated from fisheries models. This requires judgement as to the applicability of a model, and some examples are given of potential errors resulting from uncritical use of "generic" as opposed to specific models fitted to indicator series and used to generate RPs. The role of judgement, by either scientists, managers, or stakeholders, in deciding on RP values and agreeing on them with stakeholders is essential to a precautionary management framework that will be respected in practice. Given the usual inadequate data and research backing, empirical RPs based on "best available evidence" whether historical or biologically based cannot be excluded from use.

Practical experience with the formal use of multiple indicators in fisheries management for finfish or invertebrates is still limited, but the importance of monitoring a range of ecosystem and fishery characteristics and the need for further advances in this area are highlighted. Achieving a degree of redundancy by the use of multiple indices may help avoid errors resulting from basing decisions on only one or two indicator series. The systematic use of indicators has been approached here from the perspective of their incorporation in a decision-making framework. Two approaches are mentioned: the various (D)PS(I)R frameworks and the TL approach. Both can be used simply for monitoring or can be incorporated into fisheries control laws as functional components of the fisheries management cycle. The low precision possible in most fisheries information gathering structures calls for the definition of ranges of indicators corresponding to safe, uncertain, and unsafe conditions, and the colour convention used in the TL approach to management makes it clear that if indicator variance is high, it is not a question of directing the fishery onto "optimal points" as defined by a

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TRP but of avoiding dangerous or uncertain zones, which are always likely to be poorly defined.

The growing concern with ecosystem management of fisheries resources requires that monitoring also be focussed on likely interactions with other members of the ecosystem. Notable here might be key predator and prey species, competitors, parasites, and biotic habitat components providing cover or settlement surfaces. Impacts of changes in environment affecting recruitment and bottom-up effects will need to be distinguished by the use of appropriate indicators from top-down effects. Environmental, abiotic, and biotic factors have important implications for fishery management policy, especially with regard to stock recovery, and as an integrated ecological approach to monitoring for all resources is required, this may be achieved in a cost-effective manner.

Recent experience with recovery of depleted fish stocks suggests that specifically defined fishery recovery plans will require backing from an appropriate information gathering function, with nondiscretionary management actions taken when indicators approach values preagreed to with stakeholders. These RPs, and the mandatory responses that they trigger, need to be incorporated into an overriding legislation. Work to date on stock recovery suggests that concentrating on two or very few indices such as fishing mortality and (spawning) biomass is not likely to adequately track stock recovery if regime and ecosystem changes are underway. The use of indices of productivity and, for semisessile or sessile invertebrates, indicators monitoring local density within metapopulations will be necessary.

Examples are given of how the apparently less precise or quantitative approaches discussed here may be incorporated into a system of management rules that constrains possible management choices. Combining a monitoring system of multiple indicators with a response or quota change rule, such as the consideration matrix of the FRCC, or logical statements based on indicator values, as suggested by Shepherd et al. (2001) and Halliday et al. (2001), ensures that precautionary considerations are incorporated into management decision-making. This approach can be criticized because it is less "scientific", but it nonetheless offers the potential of maintaining the fishery ecosystem within a sustainable range of conditions, is easily understandable to stakeholders, and can be extended to monitor socioeconomic performance and other issues of concern to the fishing industry.

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