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Scientific concepts for ecosystem-based management of marine invertebrates on Canada's Pacific Coast

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Abstract

Management to achieve sustainable marine fisheries requires consideration of habitat and multi-species interactions in addition to the present single-species population dynamics approaches. Such “ecosystem” approaches to fisheries management are now common recommendations from advisory bodies, and are becoming part of formal legislation. From a scientific basis, however, it is not clear how to implement or advise upon such ecosystem-based approaches. This paper discusses the scientific concepts and issues necessary to include ecosystem considerations into the management of Canada’s Pacific marine invertebrate fisheries resources. The critical concepts are to define the goals for “ecosystem management”; to define the ecosystem (time and space scales); to recognise large uncertainties; to identify appropriate “control levers”; and to go slow (incrementally) but start now. Approaches to applying these concepts are to explicitly include ecosystem thinking in present assessments; to develop indices of ecosystem status; to reduce destructive fishing practices; to provide an “allocation” for predators; to establish reserve or experimental areas; and to develop models to help focus research. A central point is that “ecosystem management” can not mean the management of ecosystems; rather it means the management of human activities with marine ecosystems.

Résumé

En plus des approches actuelles de l'étude de la dynamique des populations par espèces distinctes, un mode de gestion visant à assurer le caractère durable des pêches en milieu marin doit tenir compte de l'habitat et des interactions des multiples espèces en présence. De telles approches « écosystémiques » à la gestion des pêches sont devenues des recommandations courantes de la part des organismes consultatifs et on commence même à les incorporer aux lois. Du point de vue scientifique, la façon de mettre ces approches en pratique ou de formuler des avis n'est pas claire. Le présent document traite des concepts scientifiques et des questions nécessaires qui tiennent compte de l'écosystème dans une optique de gestion des invertébrés marins du Pacifique canadien. Les démarches cruciales à entreprendre sont d'établir les objectifs d'une « gestion écosystémique », de définir l'écosystème lui-même (échelles temporelle et spatiale) de cerner les sources principales d'incertitude ainsi que d'identifier les « modes de contrôle » convenables, puis d'envisager une progression lente (graduelle) mais de commencer dès maintenant. La mise en pratique de ces concepts comportera l'inclusion de la notion d'écosystème aux évaluations actuelles, la conception d'indices pour décrire le statut des écosystèmes, la réduction des pratiques de pêche destructives, la prise compte des prédateurs, la création de réserves ou de zones expérimentales et l'élaboration de modèles pour guider les recherches. L'essentiel est de réaliser que le terme « gestion écosystémique » ne signifie pas la gestion des écosystèmes mais plutôt la gestion des activités humaines en fonction des écosystèmes marins.

Objective and Rationale:

Development of ecosystem approaches to managing natural systems is a huge problem, with great needs for detailed research. The 1997 Canada Oceans Act and other bodies charged with making fisheries management recommendations (e.g. the Fisheries Resources Conservation Council), have strongly endorsed application of ecosystem approaches to manage marine living resources. Implementing these recommendations, however, presents several problems. There is no concensus on what ecosystem processes to consider, how to index these, and how to develop management tools to operationalise these concepts. The objective of this document is to begin to develop the scientific concepts and approaches necessary to include ecosystem considerations in the management of Canada's Pacific marine living resources. The particular focus on invertebrates is driven by the belief that including at least some ecosystem concepts into present single species assessment and management procedures can be accomplished more quickly than developing and establishing new procedures which take holistic approaches to consider entire ecosystems. The general concepts, however, will be applicable to the management of any living marine resource, and could eventually lead to elimination of the apparent management separations between invertebrates and finfish species.

The Need for Ecosystem Approaches

Present fisheries assessment and management approaches are predominantly focused on single species, and are usually conducted without regard to broader system processes. The production of fish resources, however, depends on suitable habitats and the cycling of energy and nutrients through the ecosystem. Sustainable fisheries cannot be conducted simultaneously on a predator <u>and</u> its prey, however, without considering the joint impacts of these fisheries (i.e. the reduction of a key prey species is likely to impact production of the predator). Commercial fisheries resources do not exist in a vacuum, but depend on suitable habitats and the cycling of energy and nutrients through the ecosystem. Overfishing of oysters in Chesapeake Bay may have lead to habitat and water quality changes (Rothschild et al. 1994) that resulted in explosions of ctenophores (NRC 1999). In the Gulf of Alaska, there has been a dramatic shift from a shrimp-dominated system prior to 1970 to a flatfish and gadoid-dominated system since the late 1980's (Anon. 1999). Although this shift coincided with changes in ocean conditions (Anon. 1999) it also occurred simultaneously with heavy fishing of successive populations of crustaceans (Orensanz et al. 1998), which may have reduced the ability of the crustacean populations to respond to the changing ocean conditions. On Georges Bank, excessive exploitation rates are the proximal causes for declines in the once abundant gadoid and flounder species, resulting in their replacement by sharks and skates (Fogarty and Murawski 1998). Those elasmobranchs prey on juvenile stages of groundfishes, resulting in feedbacks which will tend to maintain the system in its new state.

A shift in landings from long-lived, high trophic level species (i.e. groundfishes) to short-lived, lower trophic level invertebrates and planktivorous pelagic fishes has been described recently, which Pauly et al. (1998) have termed "fishing down the food web" and Steneck (1998) has called “fishing the trophic cascade”. The concept is that fishing of higher level predators releases the predation pressure on their prey species, which are usually invertebrates and pelagic fishes. As these lower trophic level species increase in abundance, they come to the attention of the fishing industry and become target species themselves. The results are increasing catches at first, followed by stagnation or declining catches as these species become exploited (Pauly et al. 1998). There is some evidence that such a chain of events may have occurred with sea urchin (Sala et al. 1998) and cephalopod (Caddy and Rodhouse 1998) fisheries.

Precautionary approaches to managing fisheries also recognise the need for information on the broader fish-production system and its integration into management considerations (FAO 1995; Perry et al. 1999). This suggests a need for integrating multi-species scientific and management models; however, for most invertebrate populations in B.C. the data are insufficient to develop single species models, and therefore certainly insufficient to produce multi-species models. For such data-poor species, science in support of fisheries management must use as much as possible of the limited available information, and ecosystem linkages, although poorly understood, are a potential major source of information.

Brief Review of Literature on Ecosystem Considerations in Management

Recent recognition of the global problems of sustaining high fishery yields, and increasing understanding of conservation principles (mostly in terrestrial systems, NRC 1996) have lead to several reviews which identify issues in the management of marine ecosystems. Larkin (1996) suggested that ecosystem management meant applying holistic approaches to resource management, centred on multi-species interactions within variable physical and biological environments. The essential components of ecosystem management are sustainable yield, maintenance of biodiversity, and protection from pollution and habitat degradation. Larkin (1996) noted that it is difficult to distinguish changes resulting from bottom-up (environmentally-driven) or top-down (fishing) effects. Appolonio (1994) suggested that marine ecosystem management should be based on five concepts:

1. fishing industries should behave as apex predators, that is, have relatively narrow prey specificity rather than being generalists, so that they will respond directly to changes in prey abundance;
2. simple reduction of fishing pressure may not be sufficient to rebuild stocks to pre-fishing abundance or composition, because of (unknown) competitive interactions among species;
3. restoration of fish communities will only be possible if based on knowledge of the community structure and interactions;
4. restoration of traditional stock composition may only be attained by active management efforts, such as targeted reductions of predators and competitors; and
5. all components cannot be maximised simultaneously.

Appolonio (1994) also suggested that the presence of long-lived species provides “structure” and “predictability” to ecosystems, and that their reduction or removal increases variability and unpredictability. NRC (1996) elaborated on this theme, noting that ecosystem responses to perturbations are structured by the cycle times or natural frequencies of their components. A system dominated by species with long cycle times should be more predictable (over longer time scales) than systems dominated by species with short cycle times.

Bohnsack and Ault (1996) identify a hierarchy of fishing effects on marine systems which progresses from single-species to ecosystem impacts (**Table 1**). The top level of "ecosystem overfishing" occurs when the reduction of components of an ecosystem impacts other components, in particular when these impacts are difficult to predict *a priori* and to monitor because the ecological connections are not well understood (NRC 1996). Introducing management approaches to deal with such hierarchical overfishing will most likely be easier (and successfully implemented) if they are incremental to existing management activities (V. Christensen 1996).

The Ecological Society of America examined the general scientific bases for ecosystem management (N. Christensen et al. 1996). They noted that management has tended to focus on short-term goals such as yield and economic maximisation, for reasons which include inadequate information on biological diversity, ignorance of ecosystem function and dynamics, the openness of ecosystems on large scales, and a perception that immediate goals outweigh the risks of possible future ecosystem damage. They define (**Table 2**) eight elements to ecosystem management, four fundamental scientific precepts for ecosystem management, and identify four steps needed to apply ecological science to management actions. In general, they conclude that ecosystem management reflects a formal acknowledgment that populations are interdependent within ecosystems, and require that these interactions be recognised to achieve management goals (NRC 1996).

The Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), established in the late 1970's, has two principles explicitly recognising ecosystem concerns: "... maintenance of the ecological relationships between harvested, dependent and related populations of Antarctic marine living resources ...", and "... prevention of changes or minimisation of the risk of changes in the marine ecosystem which are not potentially reversible over two or three decades ..." (Gulland and Garcia 1984). They have established a scientific program to detect changes in the condition, abundance, and distribution of animals in the Antarctic (CCAMLR 1999). The program concentrates on a few ("indicator") species of birds and seals which are predators on the species exploited (euphausiids and fish), and which are likely to be susceptible to changes in the availability of the harvested species. Four categories of parameters are measured on these indicator species, involving reproduction, growth and condition, feeding ecology and behaviour, and abundance and distribution. Since changes in parameters will be due to changes in food or environmental conditions, both of these are also monitored (euphausiid abundance and environmental indices). The data are analysed annually by the Working Group on Ecosystem Monitoring and Management, to identify trends and anomalous years and locations, and to try to distinguish natural changes from those due to the effects of harvesting.

Concepts for Ecosystem-based Management Approaches

Several issues and concepts can be derived from the above review of the needs and suggestions for ecosystem approaches in fisheries management.

Goals for ecosystem management

The crucial first step is to define explicit goals for ecosystem management. Such goals are often expressed as sustaining ecosystem composition, structure, and functioning (NRC 1996). What does this mean in practice, however? Should the ecosystem be managed towards maintenance of particular species compositions and abundance levels? What should the mix of species be: those considered economically important, or those considered to be ecologically significant? Detailed objectives would need to specify the species mix (Larkin 1996). Should the ecosystem simply be monitored to observe changes, and then “managed” to reduce “negative impacts”, however these might be defined? For example, goals of this nature might include the maintenance of natural assemblages (proportions) of long and short-lived species, in which greater care would be required to protect the longer-lived species. This implies that implementing ecosystem management must take a long-term view appropriate to the time scales of the ecosystems themselves, rather than the annual review of most present single-species management (NRC 1996). Sinclair et al. (1999) have proposed that ecosystem objectives for the Atlantic coast of Canada might include the maintenance of trophic level balance, the maintenance of habitat productivity, and the maintenance of biodiversity.

Definition of the ecosystem

Definition of the ecosystem for which human interactions are to be managed is the necessary second step, and it is not trivial. Some regions, such as the Strait of Georgia, may seem obvious, but particular management objectives may require smaller spatial scales (i.e. estuarine, rocky shores, sub-tidal or pelagic habitats; Levings et al. 1998). It might be argued that a “species-centred” definition of ecosystems would be more useful, such that the ecosystem would be very large for anadromous fish like salmon (including freshwater and oceanic components), smaller and entirely marine for pelagic squid and shrimp, and smaller still for benthic invertebrates like sea urchins or geoducks. Even for these species, however, should the ecosystem be defined to include the pelagic larval stages, which may persist and drift with the currents for two months before settlement? Clear definition of ecosystem management goals, and agreement on ecosystem boundaries that are consistent with these goals, are essential. For some goals, it may be sufficient to consider only those elements (species, habitat components, etc.) that directly impact the target species, whereas indirectly-related elements may need to be considered for other goals. Sinclair et al. (1999) recommend defining “administrative ecosystem areas” on the basis of stakeholder involvement and artificial administrative systems already in place (such as political boundaries). An alternative is to define the spatial scale of interest to be smaller than what is normally considered to be the ecosystem, such as the community or habitat (in the hierarchy of gene, organism, population, species, community, ecosystem, seascape; Bohnsack and Ault 1996). This would focus attention on those interactions closer and of more immediate importance to the target species, such as food, predators, competitors, and shelter.

Recognise large uncertainties

Present assessments of invertebrates in Pacific Region have large uncertainties, due to the lack of basic information. The complexities of the interactions among several species, none of which may be well known, means that managing ecosystems will have even greater uncertainties. At worst, practical implementation of whole ecosystem management in marine environments may be impossible because of: inadequate control over human activities; many key system species are not harvested commercially, and therefore are not well known; effects of the abiotic environment on species and their interactions are poorly known, because of non-linear dynamics (NRC 1996); and because of unconsidered effects such as pollution, terrestrial impacts, and climate variability (Botsford et al. 1997).

Control levers

It must be recognised that ecosystem management does not mean management of the ecosystem itself. It means, instead, control of human interactions with the ecosystem (NRC 1999; Sinclair et al. 1999). This important point recognises that ecosystems are naturally variable, and that this natural variability is basically beyond human control. Sinclair et al (1999) suggest that management tools available to meet ecosystem objectives are still the same as available now for single species management: gear restrictions, closed areas and seasons, quotas and bycatch restrictions, and trip length restrictions. The new aspects will appear in the performance indices, monitoring needs, and decision rules that take ecosystem considerations into account (Sinclair et al. 1999). Bax et al. (in press) suggest that the application of these tools also needs to change, to be directed towards “leverage points” which are particularly sensitive to human impacts. In their study off Australia, such leverage points were identified as certain benthic habitats which had strong influences on fish community dynamics and composition, but were also vulnerable to fishing.

Go slow, but go now

Fisheries management systems worldwide are single-species based and have large supporting institutional arrangements. It is unrealistic to expect these institutions to suddenly drop this single species focus and shift into long-term ecosystem approaches. More progress can be made incrementally, starting with modifications to existing approaches to include broader ecosystem concepts. Multi-species approaches are being explored elsewhere (i.e. ICES), but not in Pacific Region. However, the recent assessments of California mussels (Gillespie 1999) and goose barnacles (Lauzier 1999), and their joint recognition of the detrimental ecological impacts of harvesting these species, are valuable first steps. It is also clear that a long-term view is necessary, both to try different approaches and to wait until the consequences of ecological management actions are expressed in the system. Therefore, although there is a need to build ecosystem management approaches incrementally, there is also the clear need to start now.

Applying these Concepts

The above concepts are very general and unspecific, but do identify key issues that must be addressed when attempting to move towards ecosystem considerations in the management of marine activities. How to apply these concepts will depend to a large extent on the goals that are established for the system. The steps to operationalise these concepts are recommended to include some combination of the following activities.

Include broader ecosystem thinking in present assessments

This step can and should be started immediately. Adopting broader habitat and ecosystem thinking, and including these in a specific section of present assessments, can lead to unexpected insights and recognition of unexpected (from a single species perspective) concerns. An example is the recognition of the importance of both California mussels and goose-neck barnacles as key structural components of rocky intertidal shores, and the difficulty of harvesting them selectively (Gillespie 1999, Lauzier 1999) which lead to fisheries closures. A section on ecosystem considerations in present assessments would identify several features (**Table 3**) and would provide a nested consideration of effects, i.e. provide a longer-term ecological view nested around the short-term annual assessment concerns. The appropriate time and space scales for the species, perhaps by life history stage, would help determine the scale of its community or "ecosystem" (i.e. taking a species-centred view of ecosystem definition). Included in this section should be some estimation of the extent or importance of metapopulations and, if possible, which stocks might be sources and which might be sinks of new recruits. Obvious and significant associations with other species (predators, prey, competitors) and critical habitat conditions (Langdon et al. 1996) should also be noted. The section should include the potential habitat disruptions of fishing, perhaps ranked qualitatively from 1 for low disruption (i.e. hand-picking of sea cucumbers by divers) to 5 for highly disruptive activities such as bottom dredging. A long-term view would also be useful, and might include guesses as to the potential impacts of exotic species known to be invading the area, and the impacts of significant climate variability and change. Comparisons of this ecosystem section among individual species assessments should prove quite interesting.

Develop indices of ecosystem status

The characterisation of ecosystem status, also called "ecosystem health", has been receiving considerable attention in the literature recently (i.e. references cited in Done and Reichelt 1998), and is a major objective of the new DFO Oceans Directorate ("Marine Ecosystem Health"). Simple indicators of ecosystem state, if properly developed, can also be powerful means to summarise and explain changes (both natural and anthropogenically-forced) in complex ecosystem structure and function to resource managers and the public. Such indices are also often used to judge the effectiveness of management actions, if the indices are devised to correctly reflect the goals for management. These goals might include protection of predators when fishing occurs on prey species, such as is done in the Antarctic where bird and marine mammal populations are monitored and indexed to identify natural and fishing-induced changes. These goals may also include maintaining the natural balance (proportions) of long- and short-lived species, and/or maintaining the natural age-structure of individual exploited species; indices for these goals could be easily developed.

Done and Reichelt (1998) describe four potential marine ecosystem management goals and their performance indices (**Table 4**). These goals are (1) maintain (or shift) the system towards a "trophically-balanced" community structure (maintenance of trophic structure); (2) protection of endangered species; (3) protection of representative and unique systems (maintenance of habitats); and (4) maintenance of biodiversity. Evaluation of at least some of these indices, and creation of time series from them, is possible now using data from fisheries-independent surveys for shrimp, urchins, sea cucumbers, and clams, which would provide a range of systems from offshore demersal to sub- and inter-tidal communities.

An approach that is more detailed than trying to index the entire ecosystem, is to focus attention onto smaller organisational units such as communities, and to define functional groups of organisms within these communities. The management goal is then to maintain species diversity within the functional groups. The need to maintain species diversity within functional groups is perhaps greater than the need to maintain the species diversity of the whole ecosystem, since “the more functionally similar species there are in a community - that is, the greater the diversity within a functional group - the greater will be its resilience in responding to environmental change, if those species differ in environmental responses” (Chapin et al. 1997). Maintenance of species diversity within functional groups is more important for groups which have strong roles in controlling ecosystem structure and processes (i.e. “keystone” groups or species). **Table 5** is a first attempt to classify the exploited invertebrate species in Pacific Region into functional groups, and to identify those which may be “keystones”. Evaluating the spatial patterns of species diversity within functional groups would also provide information on locations with greater or lesser degrees of diversity, whether these locations are related to particular habitat features, and possibly the temporal variability of this diversity. Such information is important for the design and establishment of reserve and protected areas. In concluding this section, it should be noted that it may not be possible at present to actively push the whole ecosystem in the desired direction(s), but at the least these indices would serve to characterise system-wide properties and to evaluate their changes.

Reduce destructive fishing practices

The reduction / elimination of destructive fishing practices is an obvious action to conserve biodiversity and habitats, and to support ecosystem functioning. Evaluation of the impacts of destructive fishing practices is an obvious requirement, and involves comparisons with natural frequencies of destructive events, the ability of the system to resist these events, and the recovery time after the events. Destructive fishing practices include the use of noxious substances to force octopus from their dens, bottom trawling (i.e. Kaiser 1998), and the disruption of bottom habitats [Rothschild et al. (1994) describe the destructive effects of hydraulic tongs to fish oysters in Chesapeake Bay]. Bycatch, in particular during trawling, may have significant impacts to community composition, and is being evaluated for the shrimp trawl fishery off B.C. (Boutillier et al. 1999). An index of the habitat impacts of fishing can be developed similar to those in Table 5 for use in comparing these impacts among areas or to compare the relative impacts of one fishery with others in the same area. Such an index might be of the form:

$$V_{FI} = \Sigma\ (\alpha \times A \times F)$$

with V_{FI} the impact of fishing, α a qualitative index of the destructiveness of fishing practices, with 1 representing low disruption and 5 high disruption, A the area fished and F the frequency of repeated fishing in that area. The summation would take place over all species fished in an area in order to compare with other areas, or it could be dropped if fishing practices were to be compared within an area (i.e. the impacts of hand-picking by divers *vs*. prawn traps *vs.* shrimp trawling).

Provide an "allocation" for predators

Single-species assessment models may include predator mortality (non-fishing) in a simple manner, although I am unaware of any invertebrate assessments in Pacific Region which state this explicitly. This is usually considered as natural mortality and is included in an estimate of M, the instantaneous rate of natural mortality. Echinoderm exploitation rates are held low (<10%), perhaps in consideration of their low trophic status. In Atlantic Canada, it is assumed that up to 50% of fast growing species like shrimp are eaten by other species each year, however, there is no consideration of how this might vary interannually or be linked to the abundance of predators (Sinclair et al. 1999).

Establish reserve areas

Recommendations to establish reserve or marine protected areas are now common, and have become part of DFO's Oceans strategy, although there is some debate over their usefulness at meeting all potential goals (Walters 1998; Sinclair et al. 1999). Perry et al. (1999) discuss benefits and some design considerations for the roles of reserve areas in managing new and developing invertebrate fisheries. The evaluation of the spatial patterns of species diversity within functional groups proposed above would lead naturally to identification of (perhaps optimal from an ecosystem perspective) reserve locations. Another role for reserve areas, although in opposition to protected areas, would be to explore various experimental ecosystem management approaches. Since ecosystems are likely to respond slowly to management actions, the use of reserve areas to try different actions simultaneously would greatly shorten the learning curve and provide comparisons under the same climate conditions, since several management approaches could be tried simultaneously.

Develop models

Models are abstractions of the real world, and as such are extreme simplifications and conceptualisations. This is not necessarily a bad thing, as long as this fact is kept in mind when evaluating (or acting upon) model results. Large ecosystem models (i.e. Laevastu and Favorite 1981) have a reputation of not being useful for management, and models developed from concepts such as particle size spectra (Sheldon and Parsons 1967) are too general to meet the needs of individual species management. Assumptions about the likely processes driving changes in the system that are built into the model may also prevent other (unexpected) possibilities from being evaluated, such as complex ecosystem interactions (Paine 1984). The vast literature on ecosystems and fisheries models will not be reviewed here. However, the development of conceptual and mechanistic models of communities and ecosystems, tied as closely as possible to the identification of at least functional groups of keystone species (if not the species themselves), is essential to identify significant linkages and connections that may need to be explored experimentally or studied in the field.

Example Application

Sea urchins are a prime example of the keystone functional group of subtidal relatively sedentary benthic grazers (Table 5). They can control the distribution, abundance, and diversity of benthic algae by removing erect fleshy macroalgae and leaving encrusting coraline algae (Laurence 1975; Steneck 1998). Such changes can have significant negative impacts to invertebrates and finfishes, since their juveniles often use macroalgal forests as refuges from predators (Mann 1977). The abundance of sea urchins can be influenced by predators, in particular sea otters (Estes and Duggins 1995), fishes (Cowen 1983) and invertebrates (Duggins 1983; Tegner and Levin 1983). However, predator impact is not necessarily a linear relationship with the abundance of predators; it may also depend on the benthic habitat and the opportunity for urchins to escape predators by hiding in crevices (Sala et al. 1998). Obviously, habitat characteristics that may protect urchins from one predator (crevices protect against otters) may not offer protection against other predators (i.e. starfish). The impact of predation on sea urchin abundance demonstrates cascading trophic effects, in which release from predation as a result of fishing or predators can increase urchin abundance and decrease macroalgal abundances (**Fig. 1**). The increased abundance of urchins may then be noticed and increasingly harvested by the fishing industry, resulting in “fishing the trophic cascade”. This process suggests obvious ways to index the state of the system, and to monitor shifts from urchin-dominated to the urchin-absent conditions. For example, rapid surveys (perhaps aerial) for macroalgal beds, and in particular repeated surveys to monitor changes in abundance and location, could be used to index the system state and the changes in urchin populations. Fishing must be included as one of the predation processes that can alter urchin populations, but other predators would need to be considered, especially in locations with little or no fishing. In addition, sudden increases in urchin landings (at constant effort) may be an indication of cascading trophic effects, and therefore of reductions in natural predator populations.

In Pacific Region, two species of sea urchins are harvested commercially at present (green sea urchins, *Strongylocentrotus droebachiensis*; red sea urchins, *S. fransicanus*) and a third is being evaluated as a new fishery (purple sea urchins, *S. purpuratus*), with additional species common (white sea urchins, *S. pallidus,* and the deep-living urchin *Allocentrotus fragilis*). Each of these species prefer somewhat different habitats, with green urchins occurring in fast current areas, red urchins in slower currents, and purple urchins in the heavy surf zones of rocky outer shores. There can be extensive distributional overlaps among species, particularly between red and green urchins, as indicated by the overlap of recorded urchin landings by sub-areas (**Fig. 2**). Monitoring sea urchin populations as a functional group and an indicator of system state, and attempting to ensure the maintenance of species diversity within this functional group at relevant spatial scales and locations, would be perhaps an achievable activity and goal to begin developing ecosystem-based management of Pacific marine living resources.

Recomendations

1. Begin developing “ecosystem thinking” in present stock assessment activities by including a section in each assessment evaluating the habitat and ecosystem roles and relationships for the species being assessed, following the outline of Table 3; identify the functional group to which the species belongs, and whether it is likely to be a “keystone” component.

2. Develop conceptual models of the potential species and habitat interactions, with the aim of identifying potential “control levers” and predicting directions of impacts (e.g. trophic cascading effects). Use these conceptual models to identify critical but missing information which should be collected.

3. Develop indices of ecosystem state, starting with data collected during on-going fisheries-independent surveys. This requires collection of adequate information on non-target species observed during the survey, starting with Genus (preferably species)-level taxonomic identifications.

4. Interact with other single species assessments to define the relevant ecosystem boundaries (time and space scales), the appropriate functional group upon which management actions (“control levers”) and monitoring the outcomes of management actions will be based, and the overall goals for ecosystem-based management approaches. The very large uncertainties involved in developing these new methods requires adherence to precautionary approaches.

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Literature Cited

Anon. 1999. Gulf of Alaska small-mesh trawl survey, 1953-1997. National Marine Fisheries Service and Alaska Dept. of Fish and Game. Http://www.fakr.noaa.gov/trawl/index.htm

Appolonio, S. 1994. The use of ecosystem characteristics in fisheries management. Rev. Fish. Sci. 2:157-180.

Bax, N., A. Williams, S. Davenport and C. Bulman. In press. Ecosystem structure and management. In: Ecosystem Approaches for Fisheries Management. Alaska Sea Grant College Program, University of Alaska, Fairbanks, AK-SG-99-01.

Bohnsack, J.A. and J.S. Ault. 1996. Management strategies to conserve marine biodiversity. Oceanography 9(1): 73-82.

Botsford, L.W., J.C. Castilla and C.H. Peterson. 1997. The management of fisheries and marine ecosystems. Science 277: 509-515.

Boutillier, J.A., J.A. Bond and H. Nguyen. 1999. Halibut by-catch in the British Columbia shrimp trawl fishery. Canadian Stock Assessment Research Document 99/122.

Caddy, J.E. and P.G. Rodhouse. 1998. Cephalopod and groundfish landings: evidence for ecological change in global fisheries? Rev. Fish Biol. Fisheries 8: 431-444.

CCAMLR. 1999. Ecosystem monitoring. http://www.ccamlr.org/english/ecosystem_monitoring/

Chapin, F.S. et al. 1997. Biotic control over the functioning of ecosystems. Science 277: 500-504.

Christensen, N. et al. 1996. The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecol. Applic. 6: 665-691.

Christensen, V. 1996. Managing fisheries involving predator and prey species. Rev. Fish Biol. Fisheries 6: 417-442.

Cowen, R.K. 1983. The effect of sheephead (*Semicossyphus pulcher*) predation on red sea urchin (*Strongylocentrotus franciscanus*) populations: an experimental analysis. Oecologia 58: 249-255.

Done, T.J and R.E. Reichelt. 1998. Integrated coastal zone and fisheries ecosystem management: generic goals and performance indices. Ecol. Applic. 8 (Suppl.): S110-S118.

Duggins, D.O. 1983. Starfish predation and the creation of mosaic patterns in a kelp-dominated community. Ecol. 64: 1610-1619.

Estes, J.A. and D.O. Duggins. 1995. Sea otters and kelp forests in Alaska; generality and variation in a community ecological paradigm. Ecol. Monogr. 65: 75-100.

FAO. 1995. Precautionary approach to fisheries. Part I: Guidelines on the precautionary approach to capture fisheries and species introductions. FAO Fish. Tech. Pap. 350, Part I. 47 pp.

Fogarty, M.J. and S. A. Murawski. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on Georges Bank. Ecol. Applic. 8 (Suppl.): S6-S22.

Gillespie, G.E. 1999. Stock assessment and management frameworks for the proposed fishery for sea mussels (*Mytlius californianus*) in British Columbia. Canadian Stock Assessment Secretariat Research Document 99/116. 43pp.

Gulland, J.A and S. Garcia. 1984. Observed patterns in multispecies fisheries, p. 155-190. In: R.M. May (ed). Exploitation of Marine Communities. Dahlem Konferenzen 1984. Springer-Verlag, Berlin.

Kaiser, M.J. 1998. Significance of bottom-fishing disturbance. Conserv. Biol. 12: 1230-1235.

Laevastu, T. and F. Favorite. 1981. Holistic simluation models of shelf-seas ecosystems, P. 701-727. In: A.R. Longhurst (ed.). Analysis of Marine Ecosystems. Academic Press, London.

Langdon, R.W., R.S. Steneck, V. Goceitas, F. Juanes and P. Lawton. 1996. The interface between fisheries research and habitat management. N. Am. J. Fish. Manag. 16: 1-7.

Larkin, P.A. 1996. Concepts and issues in marine ecosystem management. Rev. Fish Biol. Fisheries 6: 139-164.

Laurence, J.M. 1975. On the relationships between marine plants and sea urchins. Oceanogr. Mar. Biol. Annl. Rev. 13: 213-286.

Lauzier, R.B. 1999. A review of the biology and fisheries of the goose barnacle (*Pollicipes polymerus* Sowerby, 1833). Canadian Stock Assessment Research Document 99/111. 30 pp.

Levings, C.D., J.D. Pringle and F. Aitkens (eds.). 1998. Approaches to marine ecosystem delineation in the Strait of Georgia: proceedings of a DFO workshop, Sidney, B.C., 4-5 November 1997. Can. tech. rept. Fish. Aquat. Sci. 2247. 165 pp.

Mann, K.H. 1977. Destruction of kelp beds by sea-urchins: cyclic phenomenon or irreversible degradation? Helgol. wiss. Meeresunders. 30: 455-467.

NRC. 1996. The Bering Sea Ecosystem. National Academy Press, Washington. 303 pp.

NRC. 1999. Sustaining Marine Fisheries. National Academy Press, Washington. 164 pp.

Orensanz, J.M. and G.S. Jamieson. 1998. The assessment and management of spatially structured stocks: an overview of the North Pacific Symposium on Invertebrate Stock Assessment and Management, p. 441-459. In: G.S. Jamieson and A.C. Campbell (eds.). Proceedings of the North Pacific Symposium on Invertebrate Stock Assessment and Management. Can. Spec. Publ. Fish. Aquat. Sci. 125.

Orensanz, J.M., J. Armstrong, D. Armstrong, and R. Hilborn. 1998. Crustacean resources are vulnerable to serial depletion - the multifaceted decline of crab and shrimp fisheries in the Greater Gulf of Alaska. Rev. Fish Biol. Fisheries 8: 117-176.

Paine, R.T. 1984. Some approaches to modeling multi-species systems, p. 191-207. In: R.M. May (ed). Exploitation of Marine Communities. Dahlem Konferenzen 1984. Springer-Verlag, Berlin.

Pauly, D., V. Christenen, J. Dalsgaard, R. Froese and F. Torres, Jr. 1998. Fishing down marine food webs. Science 279: 860-863.

Perry, R.I., C.J. Walters and J.A. Boutillier. 1999. A framework for providing scientific advice for the management of new and developing invertebrate fisheries. Rev. Fish Biol. Fisheries 9: 125-150.

Rothschild, B.J., J.S. Ault, P. Goulletquer, and M. Heral. 1994. Decline of Chesapeake Bay oyster population: A century of habitat destruction and overfishing. Mar. Ecol. Prog. Ser. 111: 29-39.

Sala, E., C.F. Boudouresque and M. Harmelin-Vivien. 1998. Fishing, trophic cascades, and the structure of algal assemblages: evaluation of an old but untested paradigm. Oikos 83: 425-439.

Sheldon, R.W. and T.R. Parsons. 1967. A continuous size spectrum for particulate matter in the sea. J. Fish. Res. Bd. Canada 24: 909-915.

Sinclair, M.M., R. O’Boyle, L. Burke and S. D’Entrement.. 1999. Incorporating ecosystem objectives within fisheries management plans in the Maritimes Region of Atlantic Canada. ICES CM 1999/Z:03. 20pp

Steneck, R.S. 1998. Human influences on coastal ecosystems: does overfishing create trophic cascades? Trends in Ecol. and Evolution 13: 429-430.

Tegner, M.J. and L.A. Levin. 1983. Spiny lobsters and sea urchins: analysis of a predator-prey interaction. J. Exp. Mar. Biol. Ecol. 73: 125-150.

Walters, C.J. 1998. Evaluation of quota management policies for developing fisheries. Can. J. Fish. Aquat. Sci. 55: 2691-2705.

Table 1. A hierarchy of overfishing effects as proposed by Bohnsack and Ault (1996).

Growth overfishing	• catching too many small fish
Recruitment overfishing	• catching too many adult fish
Genetic overfishing	• catching too many fish with particular traits
Ecosystem overfishing	• removal of “keystone” species that can disrupt ecosystem structure and functioning

Table 2. Ecological Society of America recommendations for developing ecologically-based living resources management approaches (N. Christensen et al. 1996).

Elements	Sustainability	focus on intergenerational sustainability
	Goals	establish measureable goals for processes and outcomes necessary for sustainability
	Research	understanding through research
	Complexity	diversity and structural complexity strength ecosystems and provide resilience to change
	Dynamics	ecosystems will change due to their own processes
	Scale	ecosystem operate over broad space and time scales
	Humans	humans are integral components of ecosystems
	Adaptability	since ecosystem knowledge is incomplete, management approaches must be viewed as hypotheses
Scientific Precepts	Spatial and temporal scales are critical	boundaries defined for some ecosystem components are not likely to apply to other components
	Ecosystem function depends on structure, diversity, integrity	ecosystem management seeks to maintain biological diversity
	Ecosystems are dynamic	recognise that ecosystems are constantly changing, and that patch dynamics are crucial
	Uncertainty and limits to knowledge	unlikely events are certain to occur, given sufficient time and space; develop adaptive approaches
Actions	Define objectives	sustainability must be the primary objective
	Reconcile spatial scales	match management jurisdictions to ecosystem scales
	Reconcile temporal scales	take long-term views
	develop adaptable approaches	develop institutions that can adapt to ecosystem changes

Table 3. A checklist of habitat and ecosystem considerations that should be included into present single-species stock status analyses in order to recognise potential ecosystem interactions and impacts.

Time and space scales	• identify by life history stage, e.g. external fertilization?, planktonic larvae (duration, etc.)? • movement ranges of juveniles and adults
Metapopulation structures	• are metapopulations likely to exist? • sizes (qualitative) of aggregations or sub-populations • extent and nature (e.g. sources, sinks) of connections among aggregrations
Critical habitats	• identify habitat characteristics of high and low-density aggregrations, e.g. bottom type, sediment features, hydrographic and circulation features • what functional group is the species likely to belong to?
Species interactions	• prey, competitors, predators • obligate associations with other species (by life history stage) • wide or narrow ranges of prey or predators • is the species likely to be a “keystone”?
Potential for fishing to disrupt habitat, community, or ecosystem	• fishing techniques • impacts to other species • recovery times after disruptions
Long-term considerations	• susceptibility to exotic species (species introductions) • susceptibility to disruptions due to climate variations

Table 4. Possible indices of ecosystem status. The first four indices are from Done and Reichelt (1998), whereas the last index is an example of a form that could be used to assess and compare habitat impacts resulting from fishing activities.

Trophic structure index	$V_{ts} = \Sigma\ (c_j\ \alpha^{j-1})$	C_j = the proportion of biomass with j=0.5 for juvenile primary consumers, 1.0 for mature primary consumers, 1.5 for juvenile secondary consumers, 2.0 for mature secondary consumers, 2.5 for juvenile piscivores and 3.0 for mature piscivores; $\alpha = 10$. $V_{ts} = 0.3$ when 100% of biomass are immature primary consumers and 100 when 100% are mature piscivores.
Conservation status index	$V_{cs} = \Sigma\ (c_j\ \alpha^{j-1})$	c_j = the proportion of biomass with j=0.5 for “least concern”, 1.0 for “near threatened”, 1.5 for “conservation dependent”, 2.0 for “vulnerable”, 2.5 for “endangered”, and 3.0 for “critically endangered”. α = a constant (10).
Species composition index	$V_{sc} = \Sigma\ (c_j\ \alpha^{j-1})$	c_j = the proportion of colonies, plants, or bottom cover with j=1 for common, 2 for rare, 3 for previously unreported. α = a constant (10).
Bioconstruction index	$V_b = \Sigma\ (a_i\ \ m_i)$	a_i = age class *i* (in years); m_i = proportion of defined area covered by individuals of age *i*.
Habitat disturbance index	$V_{FI} = \Sigma\ (\alpha \times A \times F)$	α = destructiveness of fishing activities, with 1 = low disruption and 5 = high disruption; A = area fished, and F = frequency of repeat fishing in area A.

Table 5. Possible functional groups of commercially-exploited invertebrates in Pacific Region, with potential keystone species indicated by an asterisk (*). Initial grouping of adult spatial scale and mobility is based on Orensanz and Jamieson (1998).

HABITAT GROUP	FUNCTIONAL GROUP	EXAMPLE SPECIES		
Sedentary benthic	sedentary filter feeders	barnacles	*	(structure-forming components
		mussels	*	of rockly shores)
	low-mobility benthic filter feeders	clams (geoducks)	*	(habitat modifications and sediment movements)
		scallops		
	low-mobility benthic grazers	sea urchins	*	(control of algal composition)
		sea cucumbers		
		abalone		
Mobile benthic / demersal	territorial	octopus		
	demersal	crabs		
	demersal/pelagic	shrimp (trap-fishery species)		
Highly mobile demersal / pelagics	highly migratory	squid		
	vertically migratory	shrimp (trawl-fishery species)		
	planktonic	euphausiids	*	(significant prey species)

Figure 1. Conceptual model of the trophic cascade involving the low-mobility benthic grazers functional group (e.g. sea urchins), and the effect of changes in the abundance of predators.

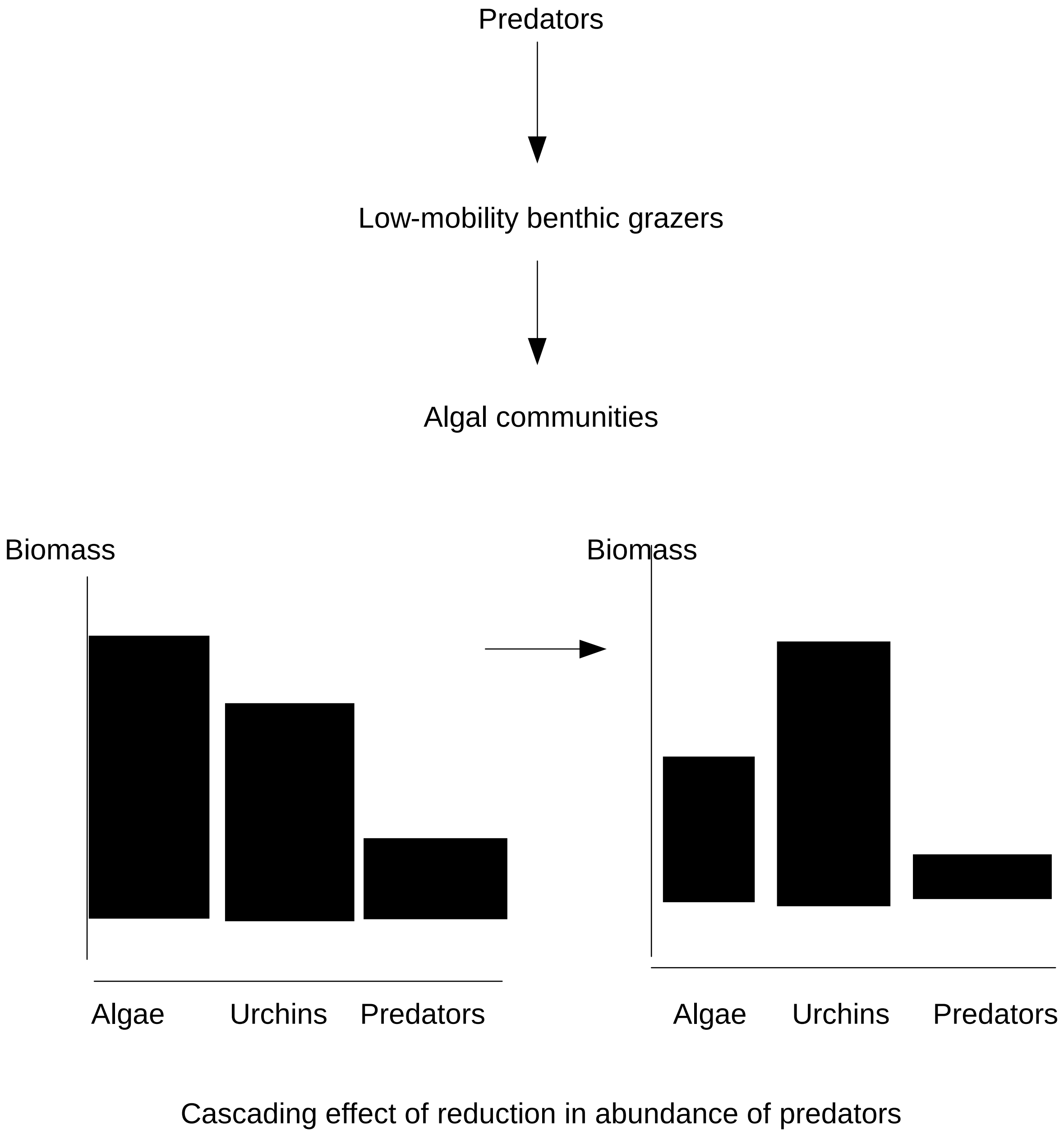

Figure 2A. Comparison of the commercial landings of sea urchins by Pacific Fisheries Management sub-area for areas 11-13 (Queen Charlotte Strait – Johnstone Strait) on the south coast of B.C. Sea urchin landings (as landed weight in kg) are expressed as a ratio (red sea urchins / green sea urchins): low values indicate sub-areas from which predominately green urchins were harvested, whereas high values indicate those in which red urchins dominated the harvest. Such a map could be used to compare “keystone” species diversity within important functional groups (such as the low-mobility benthic grazers group).

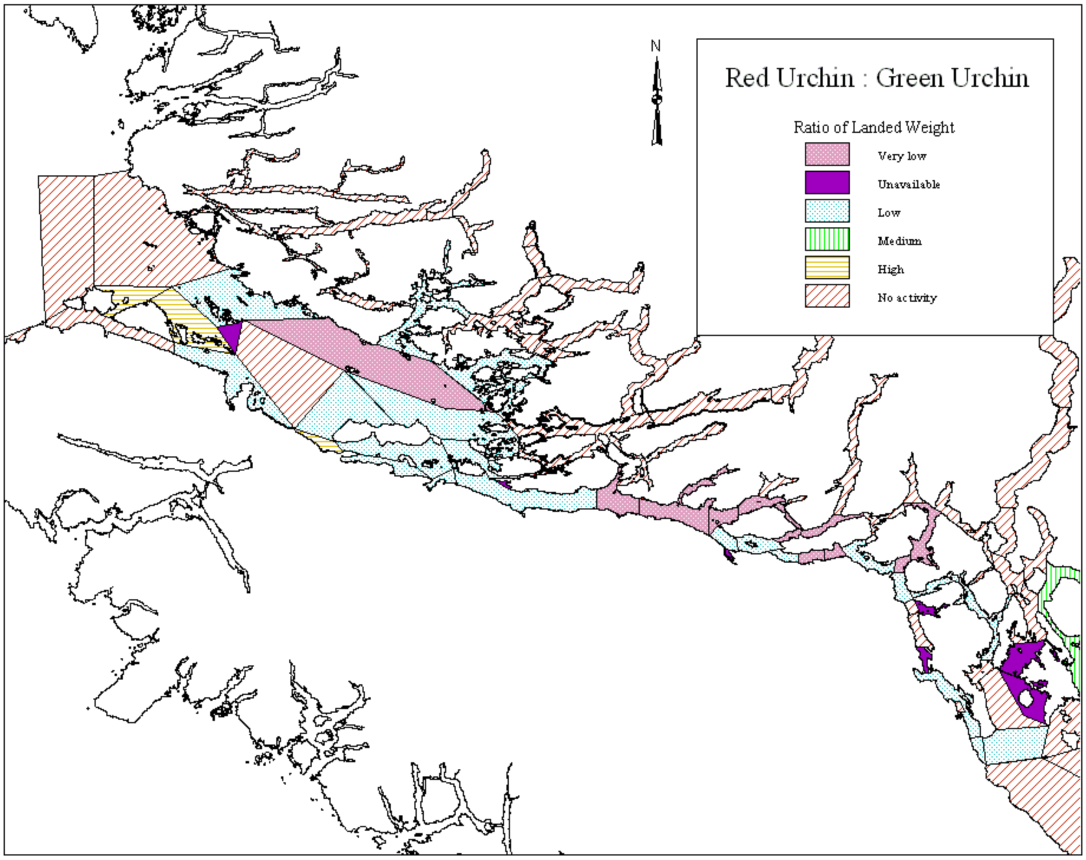

Figure 2B. Comparison of the commercial landings of sea urchins by Pacific Fisheries Management sub-area for the Strait of Georgia on the south coast of B.C. Sea urchin landings (as landed weight in kg) are expressed as a ratio (red sea urchins / green sea urchins): low values indicate sub-areas from which predominately green urchins were harvested, whereas high values indicate those in which red urchins dominated the harvest. Such a map could be used to compare "keystone" species diversity within important functional groups (such as the low-mobility benthic grazers group).