Approach for the assessment and monitoring of marine ecosystem health with application to the *Mya–Macoma* community

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APPROACH FOR THE MONITORING AND ASSESSMENT OF MARINE ECOSYSTEM HEALTH WITH APPLICATION TO THE *MYA–MACOMA* COMMUNITY

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This report develops a framework for monitoring and assessing marine ecosystem health at the level of biological communities in the littoral zone. We initially present the concept of ecosystem health. Thereafter we describe the methodology to reach a set of indicators in four major steps. First, we identify the major issues affecting a specific community. Secondly, we identify the ecological properties likely to be affected at the different ecosystem levels. Thirdly, we suggest possible indicators of these properties for the *Mya–Macoma* community. Fourthly, we explore the establishment of objectives and benchmarks necessary for an assessment. Throughout these steps, relevant definitions and concepts are presented. After this process, we evaluate the indicators suggested for the *Mya–Macoma* community according to a set of criteria and propose a series of indicators in view of each major issue affecting this community (fisheries, contamination, habitat change, and climate change). The process emphasizes the need for a prioritization at every step, consequently limiting the number of parameters needed for surveying important ecosystem components and their functions. The work presented in this report focuses on the *Mya–Macoma* community, but the approach proposed can also be applied to other communities or to another level of the ecosystem.

RÉSUMÉ

Mark, S., L. Provencher, and J. Munro. 2003. Approach for the monitoring and assessment of marine ecosystem health with application to the *Mya–Macoma* community. Can. Tech. Rep. Fish. Aquat. Sci. 2491: ix + 78 pp.

Ce rapport élabore une méthodologie pour le suivi et l'évaluation de la santé de l'écosystème au niveau des communautés biologiques de la zone littorale. Nous présentons au départ le concept de santé de l'écosystème. Ensuite, nous décrivons la méthode permettant de produire une suite d'indicateurs, suivant quatre étapes principales. Premièrement nous identifions les problématiques principales affectant une communauté spécifique. Deuxièmement nous déterminons les propriétés écologiques susceptibles d'être touchées à différents niveaux de l'écosystème. Troisièmement nous suggérons des indicateurs potentiels de ces propriétés pour la communauté à *Mya–Macoma*. Quatrièmement, nous explorons l'établissement d'objectifs et de points de référence nécessaires pour une évaluation. Au cours de ces étapes, des définitions et des concepts pertinents sont présentés. Suite à cet exercice, nous évaluons les indicateurs suggérés pour la communauté à *Mya–Macoma* selon un ensemble de critères et nous proposons une suite d'indicateurs pour chaque problématique principale affectant cette communauté (pêche, contamination, altération d'habitat et changement climatique). Le processus fait ressortir la nécessité d'établir des priorités à chaque étape, limitant par conséquent le nombre de paramètres nécessaires pour suivre les composantes importantes de l'écosystème et leur fonction. Le travail présenté dans ce rapport cible la communauté à *Mya–Macoma* mais l'approche proposée peut aussi être appliquée à d'autres communautés ou à un autre niveau de l'écosystème.

EXTENDED SUMMARY

In Canada, the littoral zone of the St. Lawrence Estuary and Gulf is being strongly affected by human activities, and new management measures are needed. The *Oceans Act* is one of the means for improving the environmental conditions. In particular, the Marine Environmental Quality (MEQ) program within the *Oceans Act* is charged with the development of methods for marine quality assessment.

MEQ builds its methodology for environmental assessment on the concept of "ecosystem health." The most important views of ecosystem health are presented in the report, reflecting the ongoing debate of the concept. In our work, we interpret ecosystem health as including both structural and functional aspects of an ecosystem. "Maximum health" is therefore found under pristine conditions, and loss of structural or functional components results in reduced ecosystem health. The concept of ecosystem health implies that a minimum acceptable level of health can be determined for areas where exploitation occurs, thus ensuring the preservation of vital parts of the ecosystem.

This report develops a framework for monitoring and evaluating ecosystem health at the level of biological communities in the littoral zone. This framework can be labelled "major issues ecosystem approach." The general approach consists of two parts, reflecting basic properties within an ecosystem on one hand and human influences on the other. Among human influences, we focus on the major issues that are thought to affect a community, e.g. pollution, fisheries. The ecosystem part of the framework considers the entire ecosystem evolving around the community and is separated into biological, chemical, and physical components. The biological component is itself arranged hierarchically into four levels: ecosystem, community, population, and individual. Each level has a set of ecological properties, normally used for its characterisation in the scientific literature. For example, we use condition, growth, and reproductive capacity at the individual level and abundance, productivity, and reproduction at the population level. During an evaluation of the possible effects of a major issue, the properties of each ecosystem level are scrutinised to identify the most affected ones. For monitoring purposes, the affected properties need to be converted into easily measured features that can be recorded repeatedly and reliably over time and space. This is the role of indicators. The criteria that an indicator should fulfil are listed in the report. Moreover, each indicator should be associated with a specific objective and benchmark.

We have applied the approach to the *Mya–Macoma* community since it is the site of important issues identified by the stakeholders and managers of the littoral zone in the St. Lawrence Estuary and Gulf. The major issues affecting this community are fisheries, pollution, habitat modification, and climate change. Taking each major issue in turn, we determine which ecosystem levels and associated properties are affected. A list of possible indicators reflecting these properties is then made. A short description of each indicator is given along with available information on its use. Following consideration of criteria on indicators, we select a final set of indicators for the *Mya–Macoma* community for each of the four major issues. This indicator set needs to be tested before implementation in a monitoring program.

An advantage of the "major issues ecosystem approach" is that it recognises the need for a prioritisation among issues, ecosystem levels, properties, and indicators. We find that this approach allows keeping the overview while entering in relatively detailed matters concerning the properties and indicators. It also aims at limiting the number of indicators needed to cover a topic. The approach encourages question-driven monitoring, since one is forced to put forward hypotheses when setting objectives for each study site. The work presented in this report focuses on the *Mya–Macoma* community, but the approach proposed can also be applied to other communities or to another level of the ecosystem.

1. INTRODUCTION

Coastal areas have long played a central role for the development of human civilisation. The use of ships for trade and transport has made these areas obvious places for development of cities, as they furnish easy access to agricultural and industrial products from the inland and fishery yields from the sea. As natural habitats, littoral areas are characterised by high productivity and recognised as nutrient traps and important nurseries for offshore fish. In addition, the littoral zone contains unique natural habitats and attracts attention for that reason alone.

The many human activities in the coastal areas impose a significant pressure on the environment, including physical alteration of habitats, over-exploitation of resources, and pollution. This pressure has increased steadily as human population has become larger; and as the limits of exploitation are reached, user conflicts are frequently seen. It has become clear that careful management of coastal areas is needed to reach reasonable compromises and to ensure adequate protection.

This situation is urging many countries, including Canada, to initiate various programs, notably on integrated management of activities in the marine environment, based on an ecosystem approach. These new programs are often rooted in international documents such as the Brundtland Report (World Commission on Environment and Development 1987) and the Biodiversity Convention from Rio 1992 (Secretariat of the Convention on Biological Diversity 2001). In Canada, the *Oceans Act* that came into effect in 1997, fosters two forms of integrated management, namely the Integrated Management (IM) and the Marine Protected Areas (MPA) programs, both applicable to coastal, estuarine, and oceanic areas. To ensure that integrated management is based on the current knowledge of the ecosystem, these two programs are supported by the Marine Environmental Quality (MEQ) program. The MEQ program includes the establishment of marine environmental quality guidelines, objectives, and criteria, and the development of methods for assessment and monitoring of marine environmental quality.

The current document represents an important step in the implementation of the MEQ program in the Quebec Region of Fisheries and Oceans Canada (DFO). The approach described in the report has been developed by the regional MEQ team. It is inspired by work done within the national MEQ study group during the year 2000, by conferences on the ecological quality of the North Sea, and by the scientific literature. Due to continuing development in the national MEQ work, the present report does not necessarily reflect recent advances in the national MEQ initiative (e.g. Jamieson et al. 2001).

This work presents an approach for selecting ecosystem health indicators, considering the links between ecosystem properties on one hand and major environmental issues on the other. For any management zone, small or large, stakeholders are preoccupied with a limited number of important issues, and these issues are themselves associated with a limited number of ecosystem components and their properties. The suggested approach consists of analysing, for a given zone, the principal issues–components associations in order to extract the smallest set of indicators that represents these interactions adequately.

In Quebec, the recent development of MEQ has aimed at supporting initiatives of DFO's Coastal Management Group, taking place mainly in the littoral zone of the Estuary and Gulf of St. Lawrence. Concerns from ZIP[1] committees, stakeholders of developing IMs and MPAs, as well as DFO's habitat protection, all pointed out a number of major environmental issues in this zone. Several of these focused on the biological community populated by the soft-shell clam *Mya arenaria.* This general concern has guided the current orientation and application of the approach described in the following pages. The formal scientific name of this community is the *Macoma balthica* community (Petersen 1913, Thorson 1957). However, given the focus on *Mya arenaria* in the issues mentioned, the community has been designated as the *Mya–Macoma* community for the purpose of this report.

The aim of this report is to develop a methodology for the monitoring and assessment of marine ecosystem health in the littoral zone. More specifically, the framework is intended for assessing a biological community within an ecosystem in view of the major issues affecting it. The report introduces the concept of ecosystem health and then describes the methodology itself, in four major steps. First, the major issues affecting a specific community are identified. Second, the ecological properties of the different ecosystem levels likely to be affected by these issues are also identified. Third, research is done on possible indicators of these properties using the *Mya–Macoma* community as an example. Fourth, we explore possible comparison levels for the indicators. Throughout these steps, relevant definitions and concepts are presented. After these steps, the indicators are evaluated according to a set of criteria and a set of indicators to be used for the *Mya–Macoma* community is proposed.

2. ECOSYSTEM HEALTH

A major objective of the MEQ program is to assess the state of an ecosystem, based on the concept of ecosystem health. The following section looks first at the efforts of the scientific community to reach a common definition and the ongoing debate of the concept, and second at the interpretation chosen and applied in the present report.

2.1. A concept under development

The term "ecosystem health" has gained importance in the field of environmental management during the past two decades. Conferences and workshops have been held (Costanza et al. 1992, Rapport et al. 1995, Rapport et al. 1998), and journals focusing on ecosystem health have appeared (*Journal of Aquatic Ecosystem Health* in 1992[2], *Ecosystem Health* in 1995, and *Aquatic Ecosystem Health and Management* in 1998). This reflects on one hand a broader understanding of environmental protection, causing a shift of focus from single components, like species, towards ecosystems. On the other hand, it reflects a general agreement among scientists and managers that ecosystems change as a consequence of human impact and that it

[1] ZIP: *Zone d'intervention prioritaire*—Priority intervention zone—a program promoting a better understanding of the St. Lawrence to facilitate local initiatives for protection, restoration, conservation and sustainable development of the resources and the uses of the St. Lawrence. At present there are 14 local committees distributed along the river, the estuary and the gulf.

[2] This journal was from 1997 onwards named: *Journal of Aquatic Ecosystem Stress and Recovery.*

is desirable to reduce these impacts. It does not, however, reflect a consensus on the meaning or even the usefulness of the term "ecosystem health."

A range of definitions appears in the literature, representing different views on the general goal of ecosystem management and conservation (e.g. Karr 1993, Suter 1993, Calow 1995, and Rapport 1995). These definitions range from including biophysical, human, and socio-economic components together with definitions focusing primarily on the biophysical aspects. Even definitions focusing on a single component within a biotope can be found (Rapport 1995). Some advocate that ecosystem health should include considerations of human health and human benefits of ecosystem functioning (Calow 1995, Rapport 1995), whereas others define a healthy ecosystem as an "ecosystem that existed prior to cultural impact" (Smol 1992 in Rapport 1995). This means that while some include human well-being in the concept of ecosystem health, others wish to exclude humans from "healthy nature." Others do not fully embrace the ecosystem view when practising the concept as they maintain focus on components such as water quality. As could be expected, these diverging opinions create a vigorous debate on what ecosystem health is and on the advantages and disadvantages of using such a term. Without engaging in details of the various definitions, we briefly review some of the general arguments for and against the use of the concept of ecosystem health to illustrate the debate.

As already mentioned, one asset of ecosystem health is that it focuses on whole systems (as do similar concepts, e.g. ecological integrity) and not merely on parts of it. In addition, the health concept includes the idea of potentials, e.g. potential for recovery, which may be important to include in environmental assessments. Relevant health parameters are easily identified at the individual level, where they could be expressed in terms of potential for growth and reproduction, whereas at the population level they could be expressed in terms of reproductive capacity and maturity. In biological communities or associations, the parameters become less obvious, but they might be expressed in terms of potential for reaching higher successional stage, and in ecosystems, for higher complexity and longer food chains.

An unavoidable problem using system views is that assessments demand a norm or a standard that a state can be compared to. This is not only the case for ecosystem health, but for any concept used in ecological assessments. We need to know if and how a system deviates from what is expected, which is not easy given the complexity of ecosystems. Even within the same area, chance factors may cause two sites to be different although they seem to have exactly the same natural conditions and history. Additional problems arise from the difficulty in distinguishing variation caused by natural and anthropogenic factors. On top of that, we realise that often our knowledge of an ecosystem is insufficient even to allow determination of the range of natural variation.

One of the major arguments for using ecosystem health is the power of analogy obtained from human health science. Just as in medicine, a "diagnosis" in environmental assessment has to be made on less than perfect information. Therefore, the process of developing methods for assessing ecosystem health can perhaps benefit from the experience accumulated in health sciences (Rapport 1995).

This analogy, however, might give a false impression of the nature of environmental management and lead to unacceptable management priorities (Suter 1993). In medicine, a body part is readily replaced by an artificial one, as long as it is functional. The equivalent of replacing a worn hip would perhaps be introducing an exotic species to manage a functional problem in an ecosystem (e.g. a fish species to deal with an eutrophication problem). Even if most biologists would agree not to understand the concept this way, the association is very obvious when borrowing from health science. But as Noss (1991) states: "Our desire to manage everything is exceedingly arrogant given our ignorance of how nature works." Acknowledging this view, perhaps ecosystem health is not the term encouraging the appropriate caution with respect to "managing" nature.

Another argument for using the ecosystem health concept is that it provides a very powerful metaphor for communicating with the public (Rapport 1995). Ecosystem health creates an image intuitively understood by most people by the association with human health. The metaphor thus makes it easier to grasp concepts of "pathology, early warning signs, costs of delayed action, advantages of preventative approaches…", etc. (Rapport 1995).

Nonetheless, the value of the metaphor has been questioned by several (Suter 1993, Calow 1995, Wichlum and Davies 1995). The analogy with human health implies that an ecosystem can be viewed as an organism, a notion that is not consistent with contemporary studies. Suter (1993) points out that ecosystems have no clear boundaries, they do not have consistent structures from one example to the next, they do not develop in a consistent manner, and they do not have mechanisms to maintain homeostasis as do organisms (e.g. neural and hormonal systems). Ecosystems are not subject to natural selection and a particular state cannot be said to be good for the ecosystem, whereas good and bad for an individual can be determined in terms of fitness (Wichlum and Davies 1995). Strictly speaking, the analogy to human health is therefore invalid (Suter 1993, Wichlum and Davies 1995).

Most scientists already agree on not taking the health metaphor too far (Martinko 1995), thus it is hoped that most of the potential problems can be avoided. The use of the concept of ecosystem health has, however, gained momentum and is now being adopted by governmental agencies. We must therefore consider that the term is in a state of refinement and elaboration, and avoid confusion by stating in each case the interpretation of ecosystem health.

2.2. Ecosystem health in the present context

One of the first definitions of ecosystem health to be accepted more widely appeared following a workshop on ecosystem health (Costanza et al. 1992); it is given here in a slightly modified version (see Appendix 1):
"An ecological system is healthy if it is active and maintains its organization and autonomy over time and is resilient to stress."

This implies that key traits in a healthy system are activity, organisation, autonomy, and resilience. The healthy system should thus be productive (maintain its normal productivity) and its energy and material cycles be functioning; it should uphold its structure, not depend on energy brought in by humans, and be capable of recovery after an exposure to stress.

Although the above definition was largely agreed upon at the workshop, the definition is still open to various interpretations. Indeed, a number of interpretations exist. This means that although we choose the above definition, we have to specify the interpretation used in the present context. In order to do so, we will describe the major lines of understanding of which two are prevailing.

The first interpretation sees ecosystem health as depending mainly on the functional components and processes within an ecosystem (e.g. energy flows, photosynthesis, and decomposition), whereas the identity of specific components (i.e. species) are less important as long as the system is still functioning (Callicott and Mumford 1997, Callicott et al. 1999). From this viewpoint, health is maintained if the functional aspects of an ecosystem are maintained, which requires only the conservation of major structural components. For example, the process of photosynthesis may be performed equally well, functionally speaking, by many different plant species. In this view, some species may be expendable as long as the level of photosynthesis is maintained.

The second interpretation includes in ecosystem health all the structural components of an ecosystem as well as the processes (the structures being understood as communities and their species and populations). Following this interpretation, any loss of structure or process would result in loss of health and integrity (Karr 1993). Thus, this approach does not make a distinction between ecosystem health and ecological integrity, the latter term being defined as the integral composition of the biological elements (i.e. biological diversity) and the biological processes.

In this report we interpret ecosystem health according to the above definition, as represented by Karr (1993). This means that both structural and functional aspects need to be considered in assessing marine ecosystem health. “Maximum health” is therefore found under pristine conditions (Fig. 1) and any loss of structural or functional components results in a reduction in ecosystem health.

The standard or norm that a state is compared to is thus the state that would develop under natural conditions, i.e. natural physiographic, geographic, and climatic as well as physical and chemical conditions. In addition, the natural successional stage should be taken into account, thus considering the time the area has had to develop. However, it is often extremely difficult to distinguish between natural and anthropogenic causes bringing about a given developmental stage. Finally, although the standard is ideally found where the anthropogenic influence on the ecosystem is minimal, this level can be very difficult or even impossible to determine in practice. For this reason we may at times be forced to determine a comparison level otherwise, e.g. from an arbitrary point in a time series (TemaNord 1999).

Despite these practical problems, we believe that it is of great value to maintain the ideal reference level of ecosystem health at pristine conditions. If the scale for evaluating ecosystem health itself is continuously adjusted to fit local needs and goals, as some advocate (Calow 1993, 1995), the scale becomes more and more devaluated as ecosystems degrade (i.e. a kind of inflation occurs). This could “disguise” the degree of ecological integrity and health loss.

Rather than adjusting the scale itself on a case-by-case basis, we suggest fitting the objectives according to the local needs and goals. It is thus important to note that the notion of pristine conditions only provides the measuring stick, not necessarily the goals or objectives (Fig. 1). Whatever the comparison level, this level is not to be confused with the desired level, which depends on the political, social, or economic target for a given area. If desired at the socio-economic level, various levels of exploitation could be allowed in different areas, which means accepting a lower level of "health" or integrity within these areas.

Within the MEQ program lies the intention that no ecosystem should be exploited or affected beyond a certain "minimum acceptable level" (Fig. 1). Exploitations should be sustainable and any system should be able to recover if exploitation and pressure is ceased, given time. That is, the system should maintain its resilience. This means, for example, that the basic conditions for existence of a community should not be altered (for example, the substrate). While it is not obvious to determine such a level in general terms, a minimum acceptable level from a biological viewpoint could be determined on a case-by-case basis under these general guidelines. It is evident that the lack of knowledge will limit our ability to determine the level accurately. For example, rather small changes could alter the balance between two competitors, resulting in a new community where one competitor is excluded; this is information of a detail that we rarely have in advance. It is also important to realise that determining the minimum acceptable level for an area is context dependent, i.e. the level of exploitation in the surrounding areas should be considered in the evaluation. Often exploitation leads to local loss of species so that recovery depends on recolonisation of species from neighbouring source populations. Unexploited areas within dispersal distance that are harbouring these species should therefore be protected by giving them a different "acceptable level."

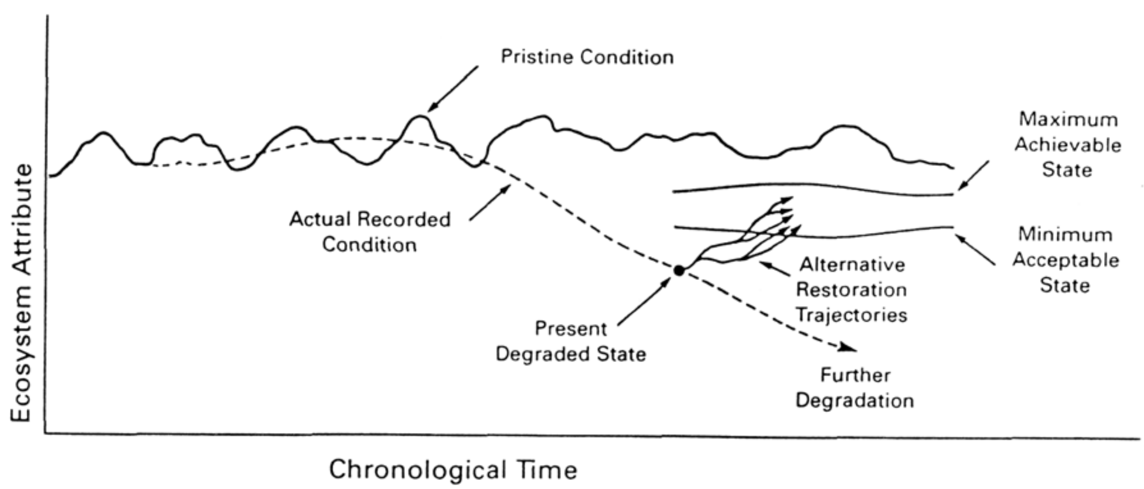

Fig. 1. Monitoring the health of an ecosystem over time (from Cairns et al. 1993, with kind permission from Kluwer Academic Publishers). The pristine condition on the figure is equivalent to our ideal reference level. At times, the actual recorded condition will have to serve as reference level, if the pristine conditions or other comparison levels are unknown. Indicated are also the minimum acceptable state and the maximum achievable state, the last indicating the best possible recovery.

One reservation that needs to be taken concerning recovery is that already the majority of communities are no longer in their pristine state. Considering the level of human impact, a complete recovery to pristine conditions may no longer be possible beyond a certain point called the maximum achievable state (Fig. 1). However, in some situations (e.g. Marine Protected Areas) the management goal may well be to achieve the maximum achievable state.

3. APPROACH FOR ASSESSING ECOSYSTEM HEALTH

The "major issues ecosystem approach" (Fig. 2) considers the ecosystem and human uses (categorised into major issues), simultaneously (Lanters et al. 1999, TemaNord 1999). The method consists of identifying parameters or variables that reflect basic ecosystem properties on one hand and major issues on the other, so as to provide a linkage between the two. This procedure emphasises the need for a prioritisation at every step, consequently limiting the parameters needed for surveying important ecosystem components and their functions.

A major issue is defined as a complex of human activities and factors that constitute a threat to one or several ecosystem elements. Examples are fisheries, pollution, and habitat modifications. By focusing on major issues, we favour detection of changes where they are most likely to occur. However, the natural factors should also be considered as they may act as confounding variables relative to the human factors. This approach may also allow for development of hypotheses concerning the direction and degree of a certain impact, steering us toward hypothesis-based monitoring as recommended by Underwood (1995, 1996). We recognise that unexpected problems might go unwarranted in this procedure, but also that it is unrealistic to cover all possible human activities and their potential impacts.

An ecosystem approach considers the components of a given system and the processes they are involved in (Likens 1992) as well as the interactions among its components. Rather than accounting for everything in the system, an ecosystem approach should aim at identifying the processes and elements that are critical for maintaining a system's characteristics.

3.1. Ecosystem organisational levels

In order to assess ecosystem health using this approach, we need to define an ecosystem and identify the various ecosystem organisational levels.

An ecosystem may be defined as *"any unit that includes all of the organisms in a given area interacting with the physical environment so that a flow of energy leads to a clearly defined trophic structure, biotic diversity, and material cycles (i.e. exchange of material between living and non-living parts) within the system"* (Odum in Cunningham et al. 1994).

Determining the limits of an ecosystem is not easy, since there is always some degree of exchange between the ecosystem and the surrounding area. Christensen and Pauly (1998) formulate that when ecosystems are properly defined "…the bulk of the trophic flow takes place between parts of the ecosystem, and this flow exceeds the flow between the ecosystem and its neighbours." In other words, the internal exchange between ecosystem components is larger than the external exchange, and the aggregations are thereby associated with each other to a higher degree than to their equivalents in the surrounding area. Nevertheless, ecosystems are often in practice defined more arbitrarily, fulfilling the needs for delimiting a study area, for example. In the MEQ framework, there is an ongoing process to delimit Large Ocean Management Areas under the *Oceans Act*. These areas will be established taking ecosystem boundaries into consideration, and a set of ecosystem objectives should be attached to each of them (Anon. 2002).

Odum's definition implies that an ecosystem may be conceptually separated into biological, chemical (material cycles), and physical components. The biological component of an ecosystem is composed of communities which themselves are composed of (sub-)populations that are composed of individuals. The physical and chemical dimensions of the ecosystem affect and are affected by the biological part, and major issues may indirectly act upon the biological dimension through the physical and chemical dimensions and vice versa. An ecosystem can thus be presented as a hierarchy of different aggregations that together present some degree of association, by energy and material flows.

When searching for possible effects of a major issue, we use the above described structural composition of the ecosystem (Fig. 2): individual, population, community, and ecosystem for the biological component of the ecosystem; plus a chemical and a physical component. Of course, the separation into such components is artificial in the sense that no biological unit exists without its physical and chemical environment.

3.1.1. Ecosystem development

Our view of ecosystems also includes the notion of ecosystem development. Although the ecosystem is no organism and its course of development is not fixed as that of an organism, some general developmental trends can be recognised. Odum (1969, 1971) presents trends to be expected in ecosystem development, representing an autogenic succession (i.e. a succession where biotic processes are stronger than interfering geochemical forces). Among others, the biomass and size of organisms tend to increase during succession whereas the production to biomass ratio (P/B ratio) tends to decrease (Odum 1971). Succession at the different ecosystem levels seems to be associated with an increase in, for example, size, numbers, complexity, and energy efficiency (biomass supported per unit energy), thus there is tendency for some kind of "growth." We will return to this aspect later, when looking for symptoms of ecosystem stress.

However, using models of ecosystem development or succession to obtain information on healthy conditions must also consider the time a site has had to develop (Schaeffer et al. 1988). A site that is at an early successional stage for natural reasons (for example, because of storms) cannot be expected to have the same community biomass as a late successional site, but it may nevertheless be in good health. As mentioned, however difficult it is, it may be very important to be able to distinguish natural and anthropogenic impacts.

3.2. Ecosystem properties

Each biological organisational level has been treated independently by different branches of ecology. From these branches, a number of properties emerge for each level, which are used to describe the biological characteristics of the individual, the population, the community, and the ecosystem levels.

We use these properties for health considerations because they are central in terms of fitness or in terms of a biological unit’s response to various environmental or biological forces. An individual’s condition depends on what kind of environment it encounters; any significant change in the environment causing a change in condition is therefore expected to ultimately have an impact on fitness. Similarly, the condition of higher ecosystem levels depends on the environment and the interactions between biological components. For example, at the population level, an impact on the survival rate would be expected if conditions become harsh. At the community level, the species composition could also change under such circumstances. The properties used to describe each biological organisational level are therefore at the same time the properties that are of prime importance in terms of response to any human impact.

For a given issue, the characteristics to look for in the chemical and physical ecosystem components depend on their interplay with the biological component. For the chemical component, we focus mainly on the properties related to nutrient cycling whereas properties for the physical component are related to the community’s environment (current, substrate, salinity, etc.).

In the following sections, we describe for each organisational level the important properties that can be impacted by various major issues (Fig. 2).

3.2.1. Individual level

Properties of importance at the individual level are in one way or another related to fitness. Fitness, defined as the lifetime reproductive success, is the ultimate parameter of significance for the individual. Naturally, fitness is affected by the physical and chemical environments, as well as biological interactions. Condition, growth, reproduction, mortality, and behavioural strategies are the principal properties related to fitness (Begon et al. 1990). At the individual level, mortality must be expressed in probabilities, which are derived from population estimates, and it is thus treated at the population level.

A. Condition

The condition or state of an individual is important for the individual’s survival and ability to grow and reproduce. A favourable or a hostile habitat affects the individual’s condition, and various condition measures are therefore useful for a general indication of both the vulnerability of an individual and the suitability of a habitat. A good condition is likely to lead to a longer life span of the individual, and Odum’s theory of ecosystem distress predicts that stress decreases life-span (Odum 1985), which affects fitness negatively. The condition of an individual may, for example, be expressed as biochemical activity (enzymes, immune response), energy reserves, fat reserves, fat percentage, or scope for growth, which is the amount of energy left for growth and reproduction once maintenance is covered.

B. Growth

Larger size is often a major determinant for an individual's success because larger individuals usually produce more offspring (although larger size also may increase certain risks, e.g. size-specific predation). Growth can be expressed in terms of size or in terms of developmental stage. Usually, it is expressed in growth rates (e.g. in weight or length per unit time) or in terms of energy allocation.

C. Reproductive capacity
Reproductive output is, along with survival parameters, directly related to fitness. Reproduction is often measured on a yearly basis. Most often, a high output is a sign of good health. However, for perennial species, the yearly output is not necessarily correlated to its lifetime output since some trade-off may favour delayed reproduction. Some plants, for example, flower mainly under stressful conditions (e.g. intertidal *Zostera marina*), in which case a high reproductive output that year would not indicate good health (Stevenson 1988). Indicators of reproductive output at the individual level may be measured directly using counts of offspring (at different developmental stages), or indirectly, using, for example, the size of reproductive organs.

D. Behaviour
Behaviour reflects the sum of environmental conditions and may change in order to escape or compensate for environmental stress. Although behaviour can be a powerful early-warning indicator under laboratory conditions, it is often less applicable in the field. However, for less mobile species, such as many endobenthic organisms, some general trends may be inferred from their movements; for example, shallower burial depth following uncovering due to digging.

E. Disease and parasites
The susceptibility of an individual to diseases and parasitic infections are also related to its condition, i.e. a poor condition tends to increase the disease frequency and/or the parasite load. This further weakens the individual and leaves less energy for reproduction and survival. Disease and parasitism also occurs in pristine populations, so the link between disease and pollution, for example, must be established, and measurements should consider the differences in frequency of disease and parasitism.

3.2.2. Population level

Populations are described in terms of abundance, structure and distribution, and their rate of growth. These measures are composed of descriptors such as abundance and biomass; productivity, reproduction, and mortality; age structure, sex ratio and genetic diversity; and range. Although we use the word population throughout the text, we may actually be dealing with sub-populations when looking at a specific community.

A. Biomass
An increase in biomass of a population means that the population size increases or that the individuals already present grow. In both cases, it indicates good conditions for the population. Measuring population size in terms of biomass may sometimes be preferred to abundance in numbers, depending on the organisms. For example, very dense and numerous assemblages of individuals such as bacteria mats are more easily measured this way. Biomass is usually measured as wet or dry mass per unit area or volume.

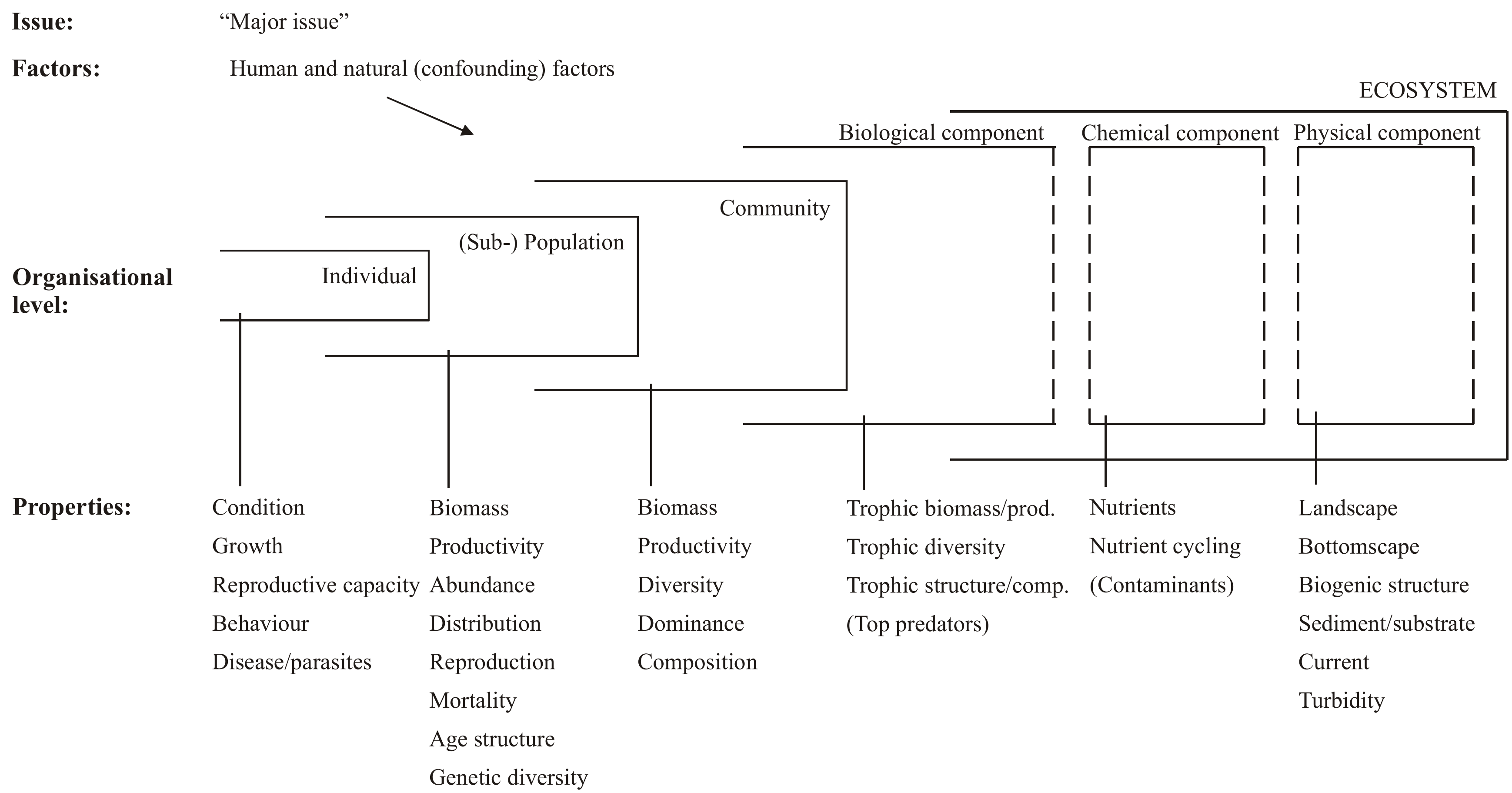

Fig. 2. Diagram of the "major issues ecosystem approach." It presents, for each hierarchical ecosystem level, the important properties that can be affected by a particular major issue. The application of the approach is illustrated in Figs. 6 to 9, where the effects of fisheries, pollution, habitat modifications, and climate change on *Mya–Macoma* communities are considered.

B. Productivity

Generally, high productivity of a population is taken as a health sign. When conditions are favourable for a population, this normally leads to higher production, either in terms of increased biomass per individual or increased reproductive output (which also can be expressed as increased biomass). However, productivity may also increase as a result of eutrophication, and high productivity is therefore not always a health sign.

Productivity is often expressed in biomass produced per unit time. Measuring productivity by measuring the change in biomass over time may not be a very good measure, however. For example, when a population is subjected to continuous predation, the proportion being predated would not be included in the productivity estimate. In these cases, indirect measures of productivity can be used, such as the reproductive or growth potential. Measures combining reproduction rates and growth rates are often recommended (Crisp 1984).

C. Abundance

In a population where the individuals are successfully surviving and reproducing, the abundance will increase. A decreasing abundance is therefore usually interpreted as a potential problem. For example, in relation to fisheries, a decrease in abundance could indicate that the exploitation rate is too high. Abundance is established by census data of various kinds (direct or indirect counts). It may be measured for the total population or as number of individuals per area (density).

There are many factors determining abundance, and measures may easily conceal underlying patterns of change. This is because abundance is made up of rates of birth and immigration and rates of death and emigration (Begon et al. 1990), so that a change in abundance could be due to a change in either one of these rates.

Population vulnerability analyses attempt to estimate the minimum viable population size (MVP). A general magic number does not really exist, since every population is faced with unique combinations of predictable and unpredictable conditions, and since viability changes continuously with population size. From the genetic viewpoint, however, a rule-of-thumb estimates that with an effective population size of 500 or several hundreds, a population should be able to avoid genetic problems (inbreeding) (Begon et al. 1990). The effective population size, which is usually lower than the actual population size, is the actual population size corrected for deviations from a “genetically ideal” population, i.e. a population with a sex ratio of 1:1, with the number of offspring varying randomly from individual to individual and with a stable population size from generation to generation (Begon et al. 1990). The genetic “magic” number is approximately an order of magnitude higher than the MVP derived from demographic considerations, so if the genetic boundary is respected, the demographic health should be ensured. More specific models estimating the probability of extinction consider environmental, catastrophic, demographic, genetic, and fragmentation forces.

D. Distribution

The spatial distribution of a population is also a necessary population descriptor. It is determined by an organism’s physical demand for the environment, food availability, and biological factors like competition or predation. Measuring distribution is thus an integrated measure of the sum of all these factors. Although it may seem relatively straightforward to measure the area covered by a population, variation over short time periods may complicate the matter (e.g. in cases of diurnal migrations), and the question of scale also needs to be considered.

The spatial distribution also furnishes data for considering a population in the perspective of landscape ecology. For example, population fragmentation may be considered and applications of fractal ecology could be particularly useful in this context (Sugihara and May 1990; Turner et al. 1999; Azovsky et al. 2000).

E. Reproduction

Under favourable conditions, reproduction tends to be more successful as a result of parents producing more offspring and/or of increased survival among the offspring. Life-history trade-offs may confound this relationship, especially for long-lived species that may choose to postpone reproduction even under good conditions. Reproduction is usually measured by the reproductive output or reproductive success per unit time, e.g. per year (Begon et al. 1990).

F. Mortality

Along with population reproductive rate and migration, the mortality rate (or its inverse measure, survival rate) determines the population abundance. Mortality rates are not straightforward to measure and are often estimated from the difference between actual numbers and expected numbers based on reproduction (immigration and emigration rates assumed to be negligible). Indicators for this property will therefore not be sought.

G. Age structure

The age structure of a population shows the result of both present and past conditions. Studying the age structure gives more detailed information than just abundance measures and provides an integration of fecundity and mortality over a time period. Although there is generally high natural variation between sites, within–site records give information on how a population develops. Major changes in age and size structure (i.e. age-abundance or size-abundance curves) could be a warning that conditions for the population in question are changing (e.g. if there are no new settlers or fewer mature individuals). The specific cause for changes is not provided by the distribution, but it might give clues to where in the life cycle the problems and causes can be found.

H. Genetic diversity

One part of biodiversity is the genetic diversity found within and between populations, i.e. within-species diversity (see Appendix). The determinants of genetic diversity are natural selection, the effective population size (N_e), and the degree of gene flow. Knowledge of genetic diversity can be obtained from genetic analysis and/or inferred from knowledge of genetic determinants such as N_e. A measure of the genetic diversity of a population is the level of heterozygosity, H. A low level of heterozygosity can, for example, indicate inbreeding depression due to low effective population size or, if the population is large, a historic bottleneck in terms of population size. Inbreeding depression is generally associated with lower reproductive success and inability to adapt to environmental changes. On the other hand, if there is sudden mixing of populations with no or little previous contact, this may result in outbreeding depression, where adaptations of a population to the local environment are lost.

Measuring genetic diversity is no simple task, and it is costly both in terms of material and man-hours. With time, the methods may become more routine, eventually making them accessible for use in monitoring programs. The best results on genetic diversity are obtained when information on both the factors affecting the genetic diversity and genetic data are available.

3.2.3. Community level

An ecological community is defined as an association of organisms present in a given environment, i.e. a group of species adapted to a specific biotope. The term is also often used to encompass species of a particular group of interest to the observer, e.g. broad taxonomic groups like fish, zooplankton, or zoobenthos (Cairns et al. 1993) that do not necessarily belong to the same biotope. In the present context, we use the first definition and use "community" to refer to the ensemble of (sub-)populations living in a characteristic biotope, e.g. the *Mya–Macoma* community. The total community may for methodological reasons be subdivided into, e.g. meio- and macrofauna communities.

When describing a community, the properties used are community biomass, community productivity, diversity, dominance, and composition (Begon et al. 1990). These properties are emerging directly from the assembly of populations, i.e. how many and which populations are present within an area and in what proportions. Communities are not entities that are as well defined as are individuals and populations, which is why community properties are usually given per unit area.

As mentioned, working at the community level means considering a whole ensemble of species. Single species approaches can be important when these are keystone species (Mills et al. 1993), umbrella species, or "flagships" (Simberloff 1998), but multi-species approaches are recommended for general environmental assessment (Cairns 1986, Wilson 1994, Smiley et al. 1998).

A. Biomass
Biomass is the weight of organisms per unit area or, in other words, the standing crop per unit area. A measure of community biomass over time provides a coarse indicator of community changes since changes in numbers or size of organisms probably will be reflected in change in total community biomass if the change is substantial enough. Biomass estimates may be useful when considered in concert with other parameters, but they are rarely useful alone due to their coarseness. At the community level, biomass is most interesting when the measures are species or group specific and can be used for community composition analysis (see below). Phytoplankton standing crop has, for example, been used for assessing trophic conditions (see 3.2.4.). Biomass is usually expressed in terms of dry organic matter (e.g. g/m^2 or g/m^3) or energy (e.g. joules/m^2).

B. Productivity
Total community productivity is the rate at which biomass is produced per unit area (net production). Like biomass, it can serve as a coarse indicator but is most useful when given with other parameters. If measured as change in biomass over time, the productivity may be hugely underestimated. This can happen, for example, when primary production is consumed

as fast as it is produced, i.e. during intense grazing (Bourgeois 1996). Measures undertaken in the laboratory may therefore be better, e.g. primary productivity measurements of a seawater sample under standard condition, or secondary productivity measurements based on female reproduction, recruitment, or growth rate of young. Productivity is usually expressed in dry or wet organic matter (e.g. g/m^2/season or g/m^3/year) or in energy (e.g. joules/m^2/day) per unit time.

C. Diversity

Diversity addresses the issue of the number of species that are present and their relative abundance. It can be measured in a number of ways, ranging from mere enumeration of species (number of species = species richness) to various indices combining the number of species and their abundance in different ways (e.g. Shannon-Wiener index, Warwick diversity index). These indices do not consider the identity of the species; thus a species may replace another with the same abundance while the index value remains the same. In most cases, though, the index value changes if species appear or disappear and if the abundance of the species changes.

It has often been assumed that degradation of an environment is followed by a lowering of species diversity. Odum's theory on ecosystem distress predicts that diversity should decline following impacts (except if the system was initially species poor—then it may actually increase). This has been shown to be true in many cases, but not all. Additionally, species diversity indices and other univariate measures are not always sensitive towards impacts of low or moderate character (Warwick and Clarke 1993, 1995).

Trophic diversity considers the number of trophic levels and also at times their relative abundance or biomass. This property is usually considered when working at the ecosystem level, but it may be useful also at the community level. Further details are given at the next organisational level.

D. Dominance

The distribution of species populations in a community is related to ecological niches. When a habitat degrades, the resource space occupied by each species will be affected according to the tolerance level of the species and the balance existing between the various components will shift. Some species, often r-selected (fast reproducing, opportunistic species) (Odum 1985) and short-lived (Rapport and Whitford 1999), become more successful and, in general terms, dominance tends to increase (Odum 1985). One way of detecting change in a community is therefore to look at the frequency distribution of species, often depicted as rank-abundance curves. Among these curves, cumulative abundance or biomass curves called K-dominance curves (see section 5.3.3) have been used for detecting deteriorated marine benthic communities.

E. Composition

Species composition changes in response to environmental change. The decline and possible extinction of habitat specialists (Odum 1985) and the invasion of exotic species (Rapport and Whitford 1999) are indications of stress, and these processes as well as more subtle ones can be analysed by looking at the ensemble of species and their abundance. This is because analysis of species composition considers both the identities of the species and their abundance (in numbers or biomass, absolute or relative). Composition analysis can therefore be a very powerful and sensitive tool for analysing complex community reactions to a complex of stressors.

Multivariate methods such as ordination and classification methods are used as tools for analysis. Ordination methods are based on (dis-)similarity matrices to clarify relations between each pair of samples (Smith et al. 1988). For example, sample pairs containing the same species with approximately the same abundance have a high similarity—up to one when the samples are identical—and zero similarity if samples have no species in common. However, the method becomes uninformative if there are many zeros in the similarity matrix because the zeros give no information on the “distance” between samples. Examples of ordination techniques are principal component analysis, detrended correspondence analysis, canonical correspondence analysis, and non-metric multidimensional scaling.

Classification methods involve grouping similar samples together in hierarchical clusters, much like taxonomic classification where similar species are grouped in genera, similar genera are grouped in families, etc. Classification is often depicted using classification trees. While ordination assumes that change occurs gradually along a continuum, classification is based on the notion that communities can be divided into relatively discrete units.

Multivariate analyses may allow an examination and indication of the causes of change through parallel analyses of environmental variables. However, this analysis is based on correlation and does not necessarily imply cause and effect relations.

3.2.4. Ecosystem level—biological component

The higher organisational level may be described as a group of interrelated communities, here referred to as the ecosystem level. At this level, we retain aspects dealing with trophic structures and energy transfer, whereas the nutrient cycling is included in the chemical component of the ecosystem. Useful parameters are trophic biomass and productivity as well as trophic diversity, structure, and composition (various functional groups and their relative abundance). In addition, species from the top trophic level are often proposed as indicators of large-scale changes.

A. Trophic biomass and productivity

Trophic biomass is the “standing crop” at the various trophic levels. Trophic productivity is an account of how much each trophic level produces per unit time. There is good evidence of major changes in the productivity of some trophic levels in relation to environmental problems (Schindler 1987). Since it is quite challenging to obtain measures of the productivity at all trophic levels, a subset of the levels may be used, for example, measures for primary production (the basis for higher level production).

There are some general trends related to ecosystem biomass and production that may be expected following stress (Odum 1985). The overall respiration tends to increase during stress, and leads to an increased respiration rate per unit biomass (Table 1). The production/respiration ratio that tends to balance ($P/R = 1$) in undisturbed ecosystems becomes unbalanced with stress ($P/R >$ or < 1). The production to biomass ratio increases. Additionally, the altered “metabolism” may result in an increased amount of unused or exported primary production (Odum 1985), i.e. the efficiency of resource use decreases. An increasing dependence on auxiliary or external energy may also be seen, so that the system becomes more open in the sense that both input and output from the system become more important (see Table 1).

B. Trophic diversity

Trophic diversity is the number of trophic levels in the system, at times taking into account their relative abundance. In distressed systems the number of trophic levels is expected to decrease, i.e. the food chains tend to shorten (Odum 1985). This may be due to reduced flow of energy at higher trophic levels or because predators are more sensitive to stress (see top predators).

C. Trophic structure and composition

One may focus on the relative contribution of the different trophic levels or of various groups within the trophic levels, e.g. feeding guilds. Examples are the ratio of deep-deposit feeders to suspension feeders and the ratio of carnivores to omnivores.

The relative contribution of pollution-sensitive to non-sensitive taxa, or simply the abundance of taxa below the sediment–water interface, may also be addressed. In the last case, a lower number of taxa would be expected in deoxygenated or polluted environments (Weisberg et al. 1997). This type of analysis is often performed at the community level, but the properties are nevertheless placed here because feeding guilds and food web considerations are traditionally considered at the highest organisational level.

D. Top predators

Top predator parameters can reveal ecosystem changes because they cover large areas and integrate the changes occurring at the lower trophic levels. They are also very susceptible to toxic contaminants because these are bio-magnified as they move up through the food web. Since these processes take time, it also takes time before any effects are revealed at the level of top predators. The taxonomic groups that include top predators are fish, reptiles, birds, and mammals.

The relevant properties at the population level of top predators are biomass, productivity, abundance, distribution, reproduction, mortality, and age structure. Since it is most valuable to acknowledge an effect before it significantly damages the top predator population, properties at the individual level are also valuable (e.g. disease frequency and behaviour). Thus, even though we aim at detecting ecosystem–level effects, the properties actually refer to the individual and population levels for top predators.

3.2.5. Ecosystem level—chemical component

The concentrations of various chemical compounds, both natural and xenobiotic, are of prime importance at this level. The term properties can only be used with difficulty at the chemical level, since we are no longer considering biological entities. What matters at this level are properties that are important in the interplay with organisms. For example, water quality parameters belong in this section.

Changes in the concentrations of natural chemicals and thus in nutrient cycling may influence the whole ecosystem (e.g. eutrophication). Concentrations of contaminants may also change ecosystems and should be considered here. Again, it is only a matter of convenience to separate this level from the lower level, since the processes of nutrient cycling are performed by biota.

A. Nutrients
The abundance of major nutrients, especially nitrogen and phosphorus, should be monitored. The simplest measures are the amount of available nutrients in the water column. However, these are highly variable in space and time. For the benthic communities, it may be more relevant to consider the concentration of nutrients in the substrate or in plant tissue, where the time variation will be reduced.

B. Nutrient cycling
It would be a tremendous amount of work to measure all links in a cycle and it is not realistic within this setting. Information on trophic biomass and productivity may be used to obtain rough estimates of the roles of some key elements in the cycle.

There are some trends that are expected in a disturbed ecosystem with respect to the nutrient cycling (Odum 1985). The nutrient turnover is expected to increase whereas the internal cycling tends to decrease. The decrease of internal cycling has also been described as an increase in horizontal transport and a decrease in vertical transport. This may lead to nutrient loss and thus a "leaky" system (Table 1).

C. Contaminants
Although this is not a property in the strict sense, concentrations of contaminants in the environment have become important to consider. In this context, contaminants are understood as natural or synthetic toxic substances that have been released into the environment by human activity (Vandermeulen 1998). Among the most important are toxic metals, pathogens (bacteria, viruses), persistent organic pollutants (POPs), and endocrine disrupting chemicals (EDCs), including, for example DDT, TBT and other pesticides, dioxins, PCBs, PAHs, and pharmaceuticals. Contaminants may be measured in the water, in the sediment, or in the tissue of organisms. The tissue concentration is an expression of exposure or pressure, since bioaccumulation is correlated with the amount of stressor in the environment but does not necessarily reveal if the contaminant causes any stress for the organism involved. Tissue concentration is valuable because it enables more effective detection at lower concentrations in the surrounding medium than direct measures, it measures the biologically active portion of a pollutant, and it integrates over time, thus including periodic bursts of contaminants that may otherwise be difficult to monitor (Depledge and Hopkin 1995).

3.2.6. Ecosystem level—physical component

The physical ecosystem is the setting around a community or ecosystem, i.e. the landscape, bottomscape, and biogenic structures, as well as the water column itself. As was the case for the chemical component, the parameters here are also of a different nature than at the biological levels. Habitat parameters provide the overall framework for aquatic life and can be described using large-scale or small-scale descriptors. Among the relevant parameters are the structure of the substrate, sedimentation, temperature, turbidity, salinity, currents, and other hydrographic measures. Obviously, some ecosystem disturbances are directly visible at this level, but changes in physical structures are not always reliable in predicting biological responses. The parameters are thus often (but not always) measured as background variables and environmental variables to be used to support interpretation of biological parameters.

3.3. Odum's distress syndrome

Throughout the previous section, we have mentioned examples of Odum's ecosystem distress syndrome. We will here briefly summarise his statements in presenting Table 1, which is a kind of summary of the preceding section. Odum (1985) puts forward a number of trends to be expected in stressed ecosystems based on his theory of ecosystem development (Odum 1969, 1971). The syndrome is built on the assumption that ecosystems react in a predictable manner when submitted to stress (Odum 1985, Rapport et al. 1985, Rapport and Whitford 1999) and based on the notion that human disturbances cause regression in the developmental stage of the ecosystem. Support for ecosystem development and for several of the expected trends can be found in the literature (Wolff et al. 1977, Schindler 1987, 1990, Christensen 1995, Rapport and Whitford 1999).

Because the symptoms of the distress syndrome tend to appear relatively late in the degradation, they do not provide for exhaustive monitoring. Therefore, additional early-warning signals are required (Rapport et al. 1985, Cairns et al. 1993), such as toxicity tests and biomonitors (Wells 1999).

4. INDICATORS AND BENCHMARKS

In a monitoring program, where the properties affected by an identified major issue should be surveyed, we must be able to measure these properties. It is therefore crucial to convert these into easily quantifiable features that can be recorded repeatedly and reliably over time and space. Many of the properties are not directly suited for this, therefore we turn to the indicator concept.

4.1. Definition of indicators

An indicator is a relatively simple measure, giving a simplified picture of a complex reality. In theory, indicators can provide an efficient shortcut to information on states and trends. The use of indicator based methods inherently holds the danger of oversimplification, but their efficiency with respect to time and resources makes them potentially very valuable. Generally, there is a trade-off between the operationality of an indicator and the power of its statement. In a holistic approach, no single indicator can provide sufficient information for us to perform an assessment.

Table 1. Ecosystem distress symptoms, mainly from Odum (1985) (points 1-18), plus some points from Rapport et al. (1985, 1998) and Rapport and Whitford (1999) ("*additional*"). The trends are organised as in Odum (1985), with the letters in parentheses referring to our organisational hierarchy. The individual, population, community, biological ecosystem, and chemical components are denoted i, p, c, e, and ch, respectively.

Odum's headings	**Trends expected in stressed ecosystems**
Energetics:	1. Community respiration increases (e) 2. Production to respiration ratio (P/R) unbalanced (> o r < 1) (e) 3. Production to biomass (P/B) and maintenance (respiration) to biomass (R/B) ratios increase (e) 4. Importance of auxiliary energy increases (e) 5. Exported or unused primary production increases (e)
Nutrient cycling:	6. Nutrient turnover increases (ch) 7. Horizontal transport increases and vertical cycling decreases (ch) 8. Nutrient loss increases (ch)
Community structure:	9. Proportion of r-strategists increases (c) 10. Size of organisms decrease, smaller biota increase (p, c) 11. Life span of organisms or parts (e.g. leaves) decreases (i, p) 12. Food chains shorten (because of reduced energy flow at higher trophic levels and/or greater sensitivity of predators to stress) (e) 13a. Species diversity decreases (if original diversity is low, the reverse may occur) (c) 13b. Dominance increases of a species (c) 13c. Redundancy of parallel processes theoretically declines (e) *Additional*: Short-lived species increase in dominance (c) Exotic species increase* (c) Possible extinction of habitat specialists (c) Disease prevalence increases (i, p)
General system-level trends:	14. Ecosystem becomes more open (i.e. input and output environment becomes more important) (e) 15. Successional trends reverse (succession reverts to earlier stages) (c) 16. Efficiency of resource use decreases (e) 17a. Parasitism and other negative interactions increase (c, e) 17b. Mutualism and other positive interactions decrease (c, e) 18. Hypothesis: Functional properties are more robust than structural properties, such as species (see Schindler 1990) *Additional*: Disruption of seasonal rhythms (c, e)

*Declining species diversity seems to facilitate invasion of exotic species (Stachowich et al. 1999).

A generally accepted definition of an indicator states that it is *"a statistic or parameter that, tracked over time, provides information on trends in the condition of a phenomenon and has significance extending beyond that associated with the properties of the statistic itself"* (Neimanis and Kerr 1996). Alternatively, the parameter is tracked not over time, but over space (i.e. between-site comparisons). In addition, a good fundamental knowledge of the behaviour of an indicator under a range of circumstances may also allow the establishment of benchmarks (e.g. minimum condition of an individual to allow reproduction).

Indicators are used at so many levels and for so many purposes that an extensive terminology has been developed in order to clarify communication. One terminology is derived from the Pressure-State-Response model (P-S-R-model), which is one of the most widely used indicator frameworks. This framework describes a cyclic feedback loop: pressure on the environment exerted by human activities, the resulting change in the environment (the state), and the societal response to the changes (OECD 1993, Anon. 1997), which then again changes the pressure. The indicators referring to each of these categories are pressure, state and response indicators, respectively. The present work focuses on the state and pressure indicators related to direct pressures on the biological community. The societal responses are not an integral part of the MEQ program, but are considered within the Integrated Management program.

Cairns et al. (1993) proposed to include three types of indicators in a monitoring program in order to produce a comprehensive and organised approach to management. These are compliance, diagnostic, and early-warning indicators. *Compliance indicators* are chosen to measure the general state of the ecosystem in relation to predetermined objectives. For example, catch per unit effort may be a compliance indicator relative to a productivity objective. Compliance indicators will usually constitute the dominant part of a monitoring program. *Diagnostic indicators* should be capable of isolating specific stress effects on compliance indicators. For example, age structure may be a diagnostic indicator for a change in catch per unit effort, revealing specific impacted age classes. Diagnostic indicators may also be chemical or physical measurements, which are then related to biological changes. Since disentangling the causes of a change in hindsight may be impossible, it is necessary to simultaneously measure some basic parameters. These background measures may then later serve as diagnostic indicators. *Early-warning indicators* should predict non-compliance with objectives before it actually shows up on the compliance indicators. These indicators inform us about subtle, sub-lethal changes (e.g. change in enzyme activity or behaviour). Including a few of such indicators in a monitoring program may allow proactive instead of reactive management.

4.2. Criteria for indicators

There are a number of criteria that an ideal indicator should fulfil (Table 2). For taxa used as indicators, additional criteria have been suggested (Table 2). It is unlikely to find indicators fulfilling all the listed criteria, although some criteria are absolutely critical, like the scientific validity and sensitivity to change. Other criteria work in opposite directions, e.g. indicators that are highly sensitive to change often work within a narrow range and thus cannot also fulfil the criteria of wide application range. It should be noted that the importance of a criteria change with the indicator type, such as compliance, diagnostic, and early-warning indicators. This should be kept in mind when assessing the fulfilment of criteria for the proposed indicators. The criteria listed in Table 2 are assumed to speak for themselves and will not be described in further detail.

Table 2. Criteria for selecting indicators.

General criteria[1]	**Criteria for indicator taxa**[2]
Properties / attributes:	*Baseline information:*
Scientific validity	Clear taxonomy
Sensitivity to change	Biology and life history known
Representative of change	Tolerance levels known
Wide application range	Correlation to ecosystems established
Tied to standards or threshold	
Integrative	*Locational information:*
	Cosmopolitan distribution
Data collection:	Limited mobility
Cost effective	
Timeliness	*Niche and life history characteristics:*
Data (historical and present) available	Early warning and functional over a range of stress levels
Easy to measure	Trends detectable
Reproducible	Low variability
Independent of the person sampling	Specialist
Simple to report	Easy to find and measure
Non-destructive collection	
	Other:
Other:	Taxa representing multiple agendas
Widely acceptable	Use of many taxa (multivariate measures)
Meaningful to social communities	

[1]adapted from Cairns et al. (1993), Nygaard et al. (1999), [2]from Hilty and Merenlender (2000)

4.3. Objectives and benchmarks

For indicators to be useful and effective, each indicator should be associated with specific objectives and benchmarks (threshold values) (Cairns et al. 1993) because the measurements then immediately suggest whether action should be taken. Benchmarks and objectives are not necessarily the same (Fig. 3). An objective is the desired level for an indicator at a specific site and depends on the societal wishes for a specific site, e.g. if the site is protected or exploited. A benchmark is a biological threshold value referring to biological limits e.g. survival, growth, and reproduction. It could also be the maximum level of contaminants tolerated by an organism. For precautionary reasons, it is desirable to set the objective well below the benchmark, e.g. at a lower concentration of contaminants.

The objective for an indicator at a specific site should comply with the overall ecosystem objective that has been set for the region. An overall objective, often stated in general terms, is thus "translated" into specific objectives associated with each indicator for each site (Jamieson et al. 2001). Realistically, these objectives can rarely be determined at the beginning of a monitoring program because the system is insufficiently known. In this case, the initial objectives could be set following the precautionary principle and adjusted at a later stage. Following the definition of ecosystem health described earlier, the ultimate reference level is pristine conditions, but the objective does not have to be that strict.

Finding benchmarks is an important, if difficult, task. Several approaches can be used to obtain benchmarks for indicators: comparison with historical data, with data from comparable but pristine sites, or with data from studies on developmental trends. Comparison with developmental trends suggests to use benchmarks like reproduction threshold levels at the individual level and a minimum viable population size at the population level. For the higher organisational levels, a developmental or successional approach is less well developed but, in the following section, we will examine the possibility of using successional trends to establish benchmarks.

Fig. 3. An indicator measuring stick. A: benchmark, B: objective, C: reference level, ideally at pristine conditions (modified from Skjoldal 1999).

4.4. Using successional trends for setting benchmarks

General successional trends along spatial gradients and through time have already been demonstrated in benthic communities (Pearson and Rosenberg 1978). These authors show general trends for abundance, species number, and biomass along an organic enrichment gradient in time and space (Fig. 4). The succession may be divided into three phases. In the first, a community of opportunist species establishes, thus leading to a peak at the abundance curve, whereas the biomass increases somewhat and the species number remains low. In the next phase, the number of opportunists decreases and other species colonise, leading to an increasing number of species and biomass. In the third phase, the number of species and the biomass stabilise at a slightly lower level with a relatively low number of individuals (Pearson and Rosenberg 1978). Recent studies have most often supported these trends (e.g. Desrosiers et al. 1990, Lu and Wu 1998, 2000, Newell et al. 1998). Another study supports the notion of regression by showing a cyclic return from a polluted state back toward a state similar to that of control sites or the original state (Clarke and Warwick 1998a).

At the ecosystem level, our limited knowledge does not allow us to set benchmarks for the time being, although a number of trends equivalent to regression in ecosystem development are expected in stressed ecosystems (Odum 1985). The trends are based on Odum's theory of ecosystem development (Odum 1969, 1971) and on the notion that human disturbances cause regression in developmental stage. Support for the regression and for many of the expected trends can be found in the literature (Schindler 1987, 1990, Christensen 1995, Rapport and Whitford 1999), but much work is still needed to further document the characteristics of ecosystem development and regression.

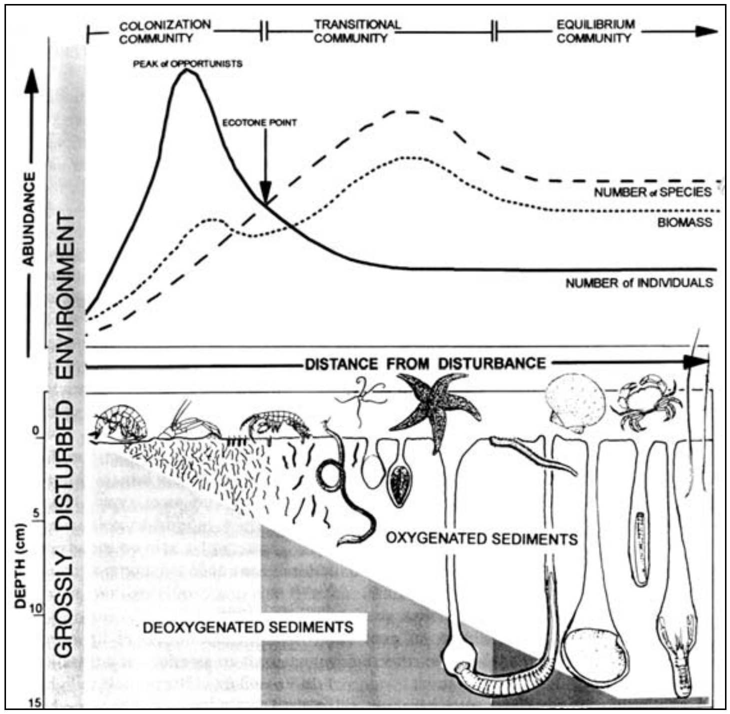

Fig. 4. Succession following organic enrichment measured in distance from disturbance (in time or space) (from Newell et al. 1998, part of their figure 2, with kind permission from the Taylor and Francis Group).

4.5. Indicator selection

We have now familiarised ourselves with the indicator terminology and criteria. Our next step is to obtain a list of possible indicators among which we can select an indicator set.

Various decision-making frameworks have been developed to assist in selecting indicators, e.g. by Smiley et al. (1998), Hilty and Merenlender (2000), and Jamieson et al. (2001)[3]. The "major issues ecosystem approach" that we propose integrates essential parts of these frameworks. It includes pressure and state indicators and makes it possible to follow the links that are made during the process between the major issues, the ecosystem properties, and the indicators. Our approach helps to keep the overview while entering into relatively detailed matters concerning the properties and indicators (Fig. 2). An ecosystem view is also encouraged throughout the process, and the priorities made along the way are clarified.

After deciding which properties the indicators should reflect, the subsequent steps are to 1) make a list of indicators associated with the properties, 2) check how they meet with selection criteria, 3) select the indicators that meet most of the criteria, and 4) complement the selection by indicators meeting criteria that were not fulfilled in the first round.

[3] Risk assessment (or threat assessment) represents an alternative or supplementary approach for evaluating an ecosystem (Suter 1993) where the focus is on probabilities of an impact. This approach has especially been used for health assessment of river and lake systems (Cook et al. 1999, Jones et al. 1999). For a recent approach to risk assessment in a multidimensional space, see Landis and McLaughlin (2000).

5. THE APPROACH APPLIED TO THE *MYA–MACOMA* COMMUNITY

In the DFO's Quebec Region, concerns expressed by ZIP[4] committees and stakeholders of developing IMs and MPAs, as well as DFO's habitat protection all identified a number of major environmental issues associated with the littoral zone. As several of these issues implicate the biological community populated by the soft-shell clam *Mya arenaria,* this community was chosen as the first for which we will develop ecosystem health indicators.

In the following sections, we will apply the major issues ecosystem approach presented in this report (Fig. 2) and explore indicators of ecosystem health for the *Mya–Macoma* community. First, the major issues for this community will be identified. The issues are dealt with one by one, using the framework to reach propositions for indicators at all relevant organisational levels. Table 1 on ecosystem stress symptoms is consulted for the ecosystem level and to some extent for the other levels.

The next steps to obtain indicators comprise checking the proposed indicators against the criteria, selecting at each organisational level the indicators that fulfil the most important criteria, and finally verifying if the set needs complementary indicators meeting other criteria. To make the approach complete, benchmarks should be set for each indicator. However, this stage is still under development and will not be applied in this report to the *Mya–Macoma* community.

5.1. The *Mya–Macoma* community

Other than *Mya arenaria* and *Macoma balthica*, the community called *Mya–Macoma* in this report also includes high numbers of polychaete worms, e.g. *Nereis virens* and *Nephtys caeca*, as well as the gastropods *Hydrobia* sp., *Littorina* sp., and other organisms. A generalised and simplified presentation of the food web involving *Mya arenaria* is given in Fig. 5, where only relationships between *Mya* and its prey and predators are depicted.

Mya arenaria (soft-shell clam) is an infaunal bivalve found in the intertidal and infralittoral area down to a water depth of approximately 10 m in the Estuary and Gulf of St. Lawrence (Biorex 1999a). In this area, the maximum size is usually around 10 cm, although larger individuals have been observed (Hawkins 1985). Along the North Shore of St. Lawrence, it takes between 5 and 9 years to reach the commercial size of 51 mm (Lamoureux 1975, Mercier et al. 1978), with variations between regions. *Mya* is a filter feeder, feeding on phytoplankton (diatoms) and zooplankton. The clam itself is buried in the substrate (preferably sand or muddy sand) down to a depth of 15-25 cm (Gosselin 1987).

[4] ZIP: *Zone d'intervention prioritaire*— ***Priority intervention zone***: a program promoting a better understanding of the St. Lawrence in order to facilitate local initiatives for protection, restoration, conservation, and sustainable development of the resources and the uses of the St. Lawrence. At present there are 14 local committees distributed along the river, the estuary and the gulf.

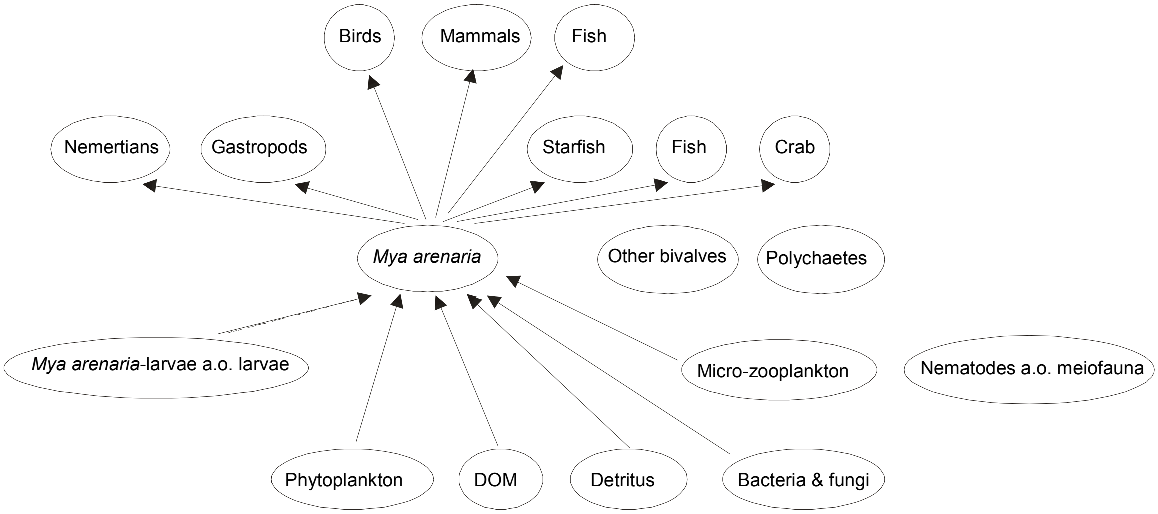

Fig. 5. Food web involving *Mya arenaria*. For simplicity, only relationships between adult/juvenile *M. arenaria* and its prey and predators are indicated. DOM = Dissolved organic matter. a.o. = and other. The figure is based on information mostly from Dame (1996).

The life cycle of *M. arenaria* (Newell and Hidu 1986) includes external fertilisation in the water column followed by a planktonic larval stage. After a couple of weeks, the larva transforms into a juvenile. Before settling definitively, the juvenile is motile for a short period, which allows it to select a suitable place for burial within a limited area. At this stage, it is around 6 mm long (Hawkins 1985).

The other dominant bivalve, *Macoma balthica*, is also an infaunal filter feeder, switching to surface deposit feeding when there is a low density of suspended particles in the water (Fish and Fish 1996, Kube et al. 1996). It lives a few centimetres below the sediment surface with its siphons extended during feeding. In the St. Lawrence region, it may grow to a maximum length of 17-19 mm (Lavoie 1970, Harvey 1990).

Nereis virens (sandworm or ragworm) burrows into the mud during the day and comes out to feed at night. It is an omnivorous scavenger, feeding on detritus, algae, and invertebrates (Fefer and Schettig 1980). *Nephtys caeca* is also a burrower, a carnivore feeding on molluscs, crustaceans, and other polychaetes (Fish and Fish 1996). *Hydrobia* is a surface deposit feeder, feeding on detritus, diatoms, and bacteria (Fish and Fish 1996), whereas *Littorina* is a herbivore, feeding on microorganisms and detritus as well as some green algae (Fish and Fish 1996). Both are epibenthic gastropods.

Among the predators of *M. arenaria* are crabs, gulls, shorebirds, sea ducks, nemerteans (*Cerebratulis lacteus*), and gastropods (*Lunatia heros*) (e.g. Emerson et al. 1990, Rowell 1992, Ambrose et al. 1998). Fish, e.g. winter flounder (*Pseudopleuronectes americanus*), also prey on clams (Vaillancourt 1982, Buckley 1989). *M. arenaria* is usually not a predator's only food source.

5.2. Major issues affecting the *Mya–Macoma* community

In the Lower St. Lawrence Estuary and Gulf, contamination, exploitation, and habitat modification have been recognised as important human activities raising concern (St. Lawrence Centre 1996, White and Johns 1997, Biorex 1999a, Biorex 1999b). As described below, the major issues more specifically associated with *Mya–Macoma* communities are fisheries, introduction of contaminants, habitat modification, and climate change.

A. Fisheries

M. arenaria populations are subject to exploitation on a commercial (using both manual and mechanical harvesting techniques) as well as a private basis (manual techniques), both of which may result in overexploitation (St. Lawrence Centre 1996). The state of these communities is often exclusively based on evaluations on the state of the *Mya* species, although the clam harvesting potentially affects not only *M. arenaria*, but also the other species making up the community.

Direct effects of the harvesting may include decreased density and distribution of a clam bed. Digging and dragging also alters the physical environment and the water–sediment interface, at least temporarily. Harvesting causes a change in size distribution of the population of *M. arenaria*, since only individuals of 51 mm and larger are removed. The excavation also damages a large portion of the small clams, leading to their death, and exposes another portion to a significantly higher risk of predation because of shallower burial following the uncovering (Emerson et al. 1990, Ambrose et al. 1998, Smith et al. 1999). In addition, the digging may directly affect the other species of the community. At the individual level, digging could result in a higher stress level due to disturbance and cost of reburial; on the other hand, it could also result in reduced intraspecific competition for space and food.

Effects at the ecosystem level are of a more unpredictable nature. Overexploitation lowers the food supply of organisms higher in the trophic chain, although top predators like gulls may initially benefit from digging as seen by the numbers of gulls following clammers. The digging may disrupt chemical processes in the sediment layers, since the layers separating aerobic and anaerobic processes are disturbed. However, the most obvious effects of the exploitation of *Mya* are at the population and community levels (Fig. 6).

B. Introduction of contaminants

Another major issue is contaminants (St. Lawrence Centre 1996) (Fig. 7). Contaminants can be divided into organic and inorganic pollutants and enter mainly with industrial or municipal wastewater (point sources) or diffuse wastewater/runoffs (e.g. agricultural runoffs and waste from private homes). The main organic substances are oil, pesticides, and sewage, including microbiological contaminants; the inorganic substances affecting the environment are mainly heavy metals and artificial or natural substances. In addition, the uncontrolled escape of genetic information into the genomes of organisms in the environment where those genes never existed before, called genetic pollution, is becoming increasingly important.

There is ample evidence that many of these contaminants have detrimental effects on individuals and that the impacts may lead to major changes at the community as well as at the ecosystem level (St. Lawrence Centre 1996). Contaminant loadings (e.g. PCB, PAH, and heavy metals) impair individual as well as population-level processes (e.g. abundance, growth, reproduction, and age structure), leading to altered community structure (Ward and Hutchings 1996, Dauer et al. 2000) and, ultimately, to altered ecosystem functioning (Schindler 1990). At the chemical level, the contaminants directly impair nutrient cycling processes, while additional organic matter from sewage may lead to increased oxygen consumption and again, changes in nutrient cycling. This influences the physical environment through changing turbidity and sediment structure.

C. Habitat modifications

Habitat modifications are also an issue of importance. These physical modifications may occur following the building of hydroelectric dams and various coastal structures (e.g. harbours, roads), the use of all terrain vehicles (ATV), and digging and land slides as well as natural storms and ice rafting. Here, we concentrate on the effect of hydroelectric dams (Fig. 8). Dams often cause a more evenly distributed outflow from the rivers throughout the year or a decrease in outflow. This especially has consequences during the spring, when a high output normally would cause an increase in nutrients, thus enabling a high primary production. The dams may thus decrease the productivity. They may also cause changes in currents, sedimentation, and salinity in the littoral areas.

D. Climate change

Finally, climate change might alter the basic conditions for the community (Fig. 9). Essentially, the temperature and current characteristics change. Although the overall trend shows an increase in the mean annual temperature, there may be local decreases (Biorex 1999a). The *M. arenaria* community is generally very tolerant of temperature change, especially in the intertidal region where the daily temperature regime changes dramatically. However, *M. arenaria* reproduction may be sensitive to overall temperature changes due to temperature– induced reproduction. Alternatively, changes in currents may lead to different erosion and sedimentation rates, altering the physical environment so as to influence the capacity for survival, growth, and reproduction for the implicated biological communities.

There are several difficulties associated with predicting the effects of climate change. It is very difficult to separate changes in the climate caused by natural and anthropogenic factors, and any measures will necessarily be related to both kinds of factors in combination. Secondly, it may be difficult to predict the direction of an expected change.

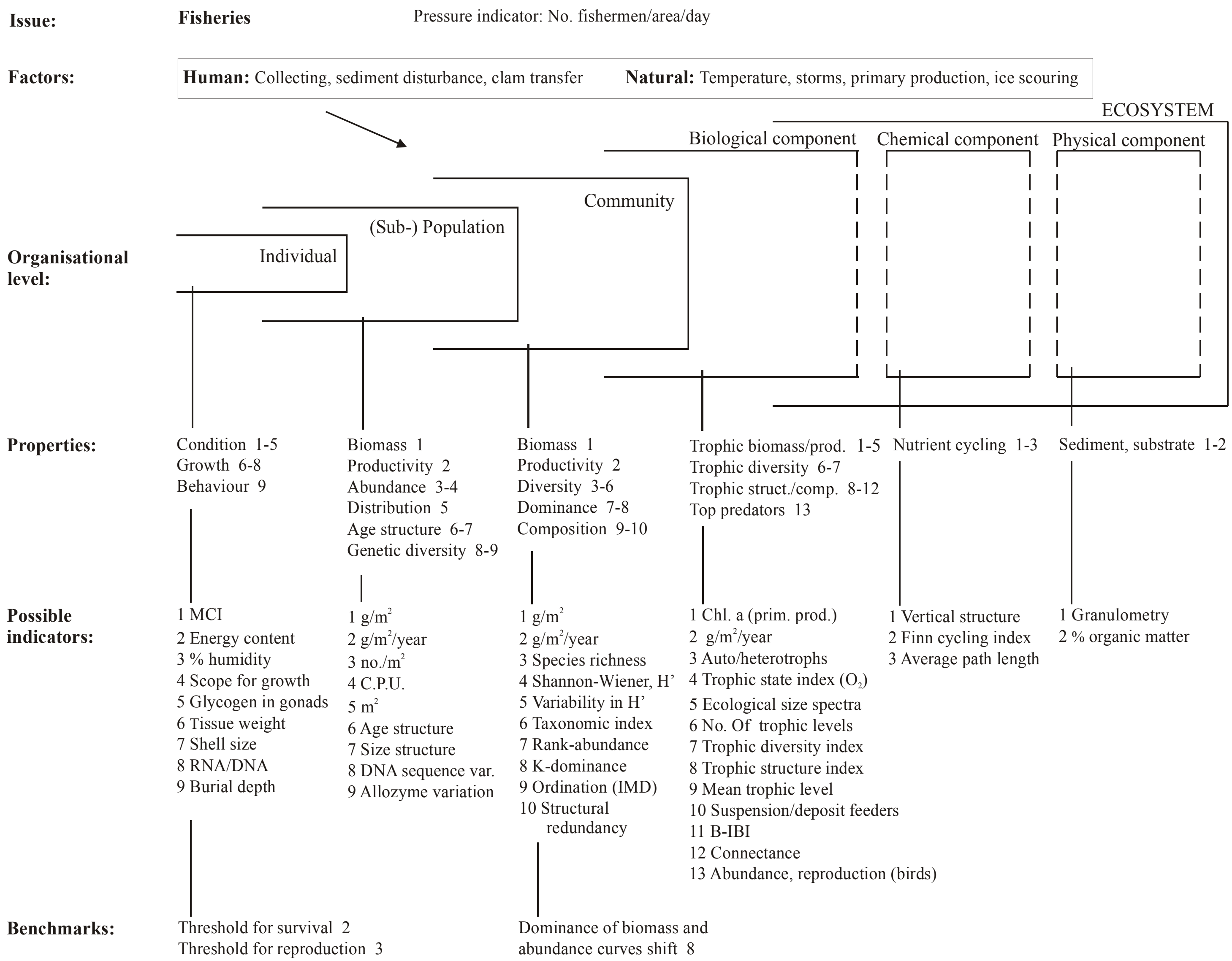

Fig. 6. The ecosystem approach considering the effects of fisheries on *Mya–Macoma* communities. The numbers to the right of properties refer to the specific indicators and benchmarks listed below. MCI = morphometric condition index, C.P.U. = catch per unit effort, IMD = index of multivariate dispersion, B-IBI = benthic index of biological integrity.

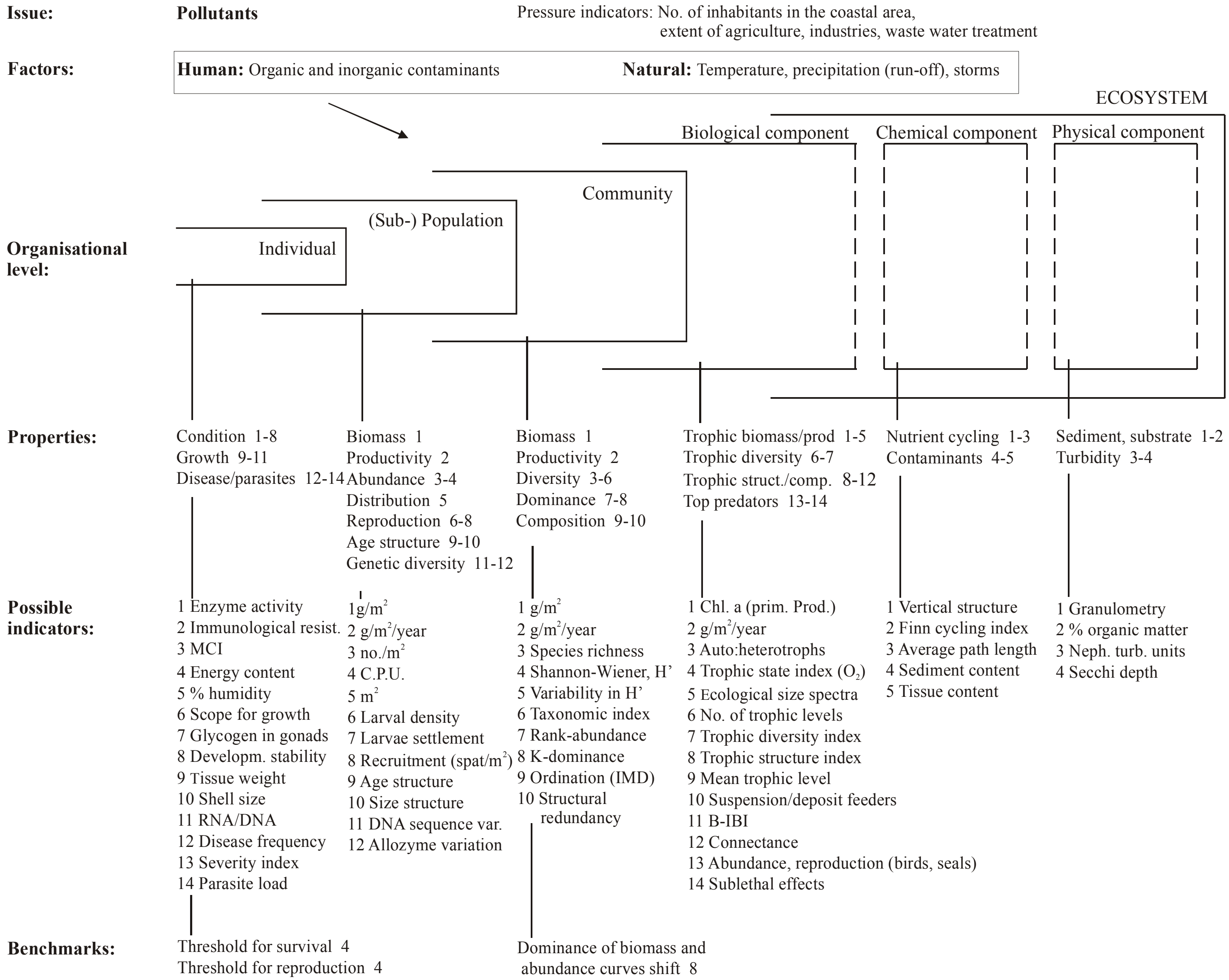

Fig. 7. The ecosystem approach considering the effects of pollution on *Mya–Macoma* communities. The numbers to the right of properties refer to the specific indicators and benchmarks listed below. MCI = morphometric condition index, C.P.U. = catch per unit effort, IMD = index of multivariate dispersion, B-IBI = benthic index of biological integrity, neph. turb. units = nephelometric turbidity units.

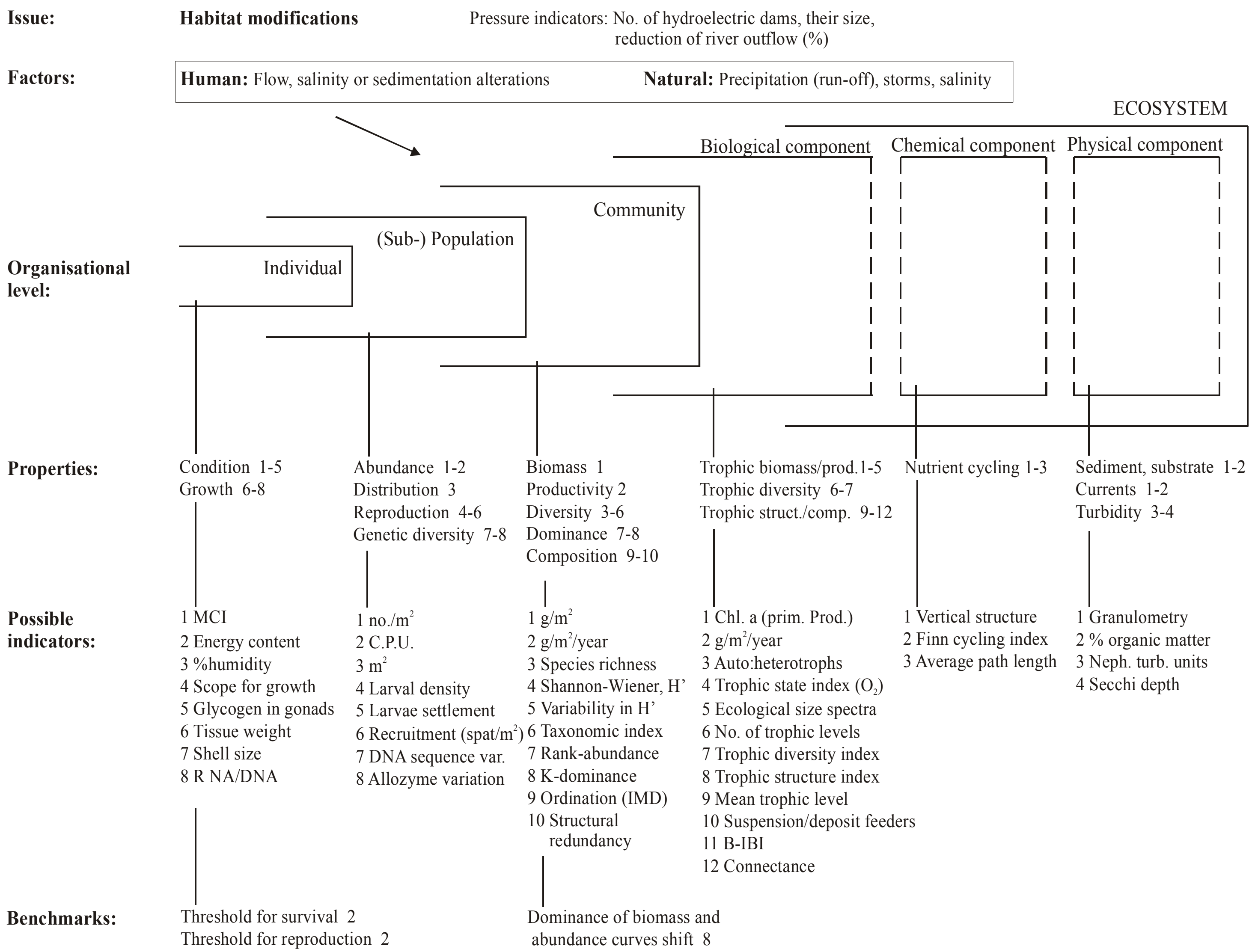

Fig. 8. The ecosystem approach considering the effects of habitat modifications on *Mya–Macoma* communities. The numbers to the right of properties refer to the specific indicators and benchmarks listed below. MCI = morphometric condition index, C.P.U. = catch per unit effort, IMD = index of multivariate dispersion, B-IBI = benthic index of biological integrity, neph. turb. units = nephelometric turbidity units.

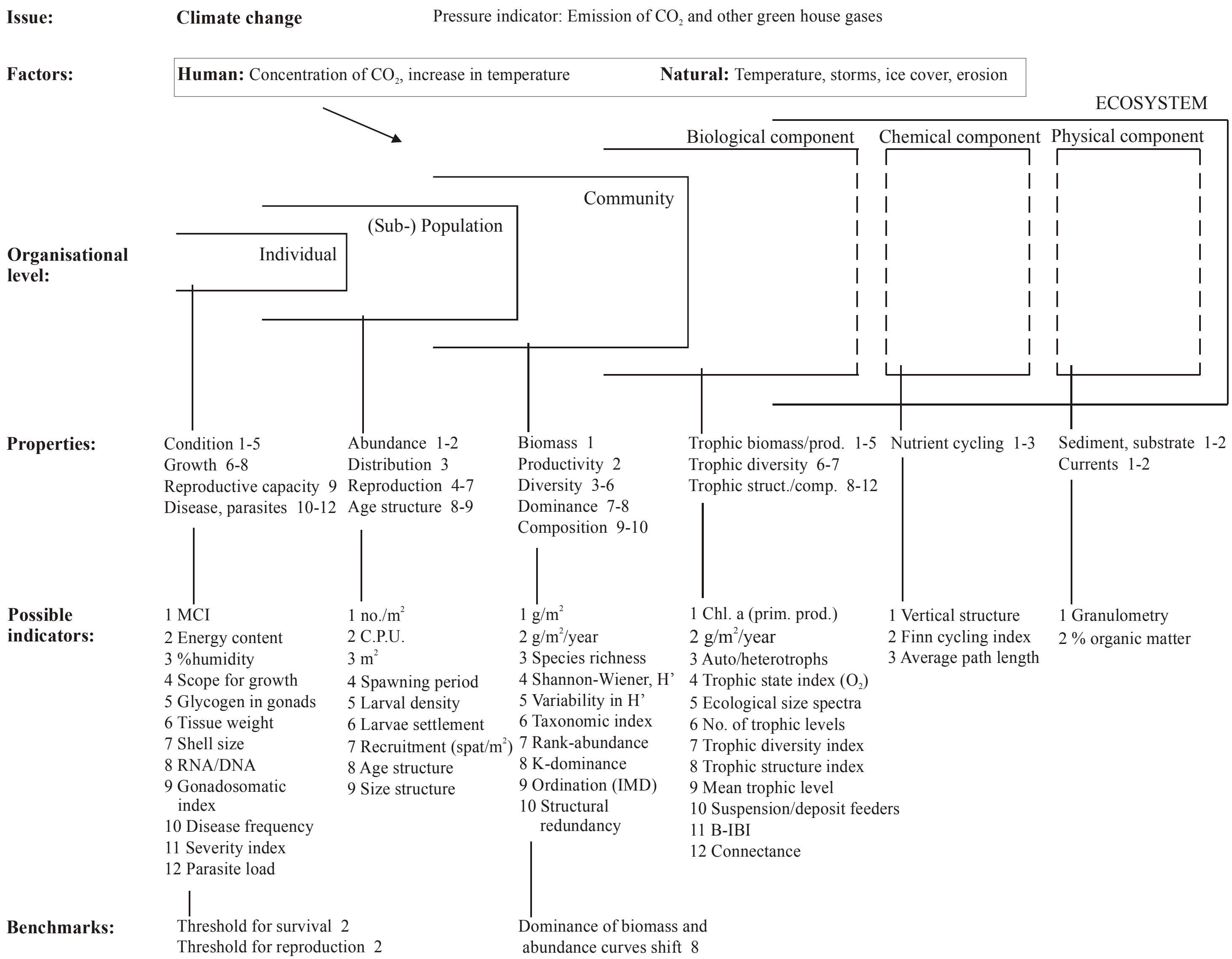

Fig. 9. The ecosystem approach considering the effects of climate change on *Mya–Macoma* communities. The numbers to the right of properties refer to the specific indicators and benchmarks listed below. MCI = morphometric condition index, C.P.U. = catch per unit effort, IMD = index of multivariate dispersion, B-IBI = benthic index of biological integrity.

5.3. Possible indicators in the *Mya–Macoma* community

For each organisational level and each property that may be affected by a major issue of the *Mya–Macoma* community, one or more indicators are listed. These are presented in the Fig. 6-9, where each figure refers to one of the four major issues identified. The indicators are described below, under the headings of the relevant properties for each organisational level. Available information on the use of an indicator is given along with the descriptions. Of course, not all possible indicators have been mentioned. When specific indicators are mentioned, they are indicators that have been used for this and similar communities.

The fisheries issue was subdivided into a circulation aspect (walking, ATVs), a collecting aspect (collection of *M. arenaria*), and a sediment disturbance aspect (digging, use of other tools, e.g. hydraulic dredge). All these are joined in Fig. 6. The organisational levels mostly affected by these activities are the population and community levels, but the individual, ecosystem, chemical, and physical levels are also involved.

Pollutants were divided into organic and inorganic compounds (Fig. 7). The effects of pollution could potentially be important at all organisational levels, but the lower level indicators should be prioritised in order to limit potential damage.

The most important habitat modification at present is probably the building and presence of hydroelectric dams (Fig. 8), but other types of constructions should be considered as well. The organisational levels to focus on are the community and ecosystem levels, and, to a lesser extent, individual and population levels.

The prime concern related to climate change is the increased mean temperature (Fig. 9). The climate change will be noticeable at the community and ecosystem levels, but the individual and population levels are also worth monitoring as they give early warning indicators.

Although each major issue gives rise to its own indicator set, the following description of the indicators considers the four major issues simultaneously. This is done to avoid repetition since several indicators are used for more than one issue. General indicators are given in **bold**, whereas indicators that are more specific or have been applied specifically to the *Mya–Macoma* community are given in ***bold-italic***.

5.3.1. Individual level

We have chosen *Mya arenaria* as the principal species for evaluating the impacts of the major issues on the properties at the individual level. *M. arenaria* is a dominant member of the community and its ecology is relatively well known (Hawkins 1985, Newell and Hidu 1986). In addition, *Mya arenaria* is the only organism being exploited in the community. Nevertheless, if other organisms in the community (for example, the other dominant species, *Macoma balthica*, or the polychaete, *Nereis*) are more sensitive to some factor, these organisms should be considered.

A. Condition

Biochemical indicators, for example **enzyme activity**, may be very powerful as early-warning indicators of stressful conditions. Such indicators are quite specific. They indicate the immediate responses of an individual, but not necessarily whether there are any long term consequences of the reaction (i.e. in terms of fitness). When an early warning indicator reveals an impact, it is thus often unknown if there is going to be an impact on the higher organisational levels.

In a study following an experimental oil spill, the activity of the physiologically important enzyme, ***glucose 6-phosphate dehydrogenase*** (G6PD), was elevated in *Mya arenaria* (and *Mytilus edulis*). The elevated activity persisted for at least nine weeks after the spill, even after the hydrocarbon level in the sediment had declined, but was normal 10 months after the spill (Gilfillan et al. 1984). The G6PD enzyme catalyses an oxidation/reduction reaction that is important for the production of both DNA and RNA and for maintaining adequate levels of intracellular NADPH. This enzyme could thus react to other stressors as well. Enzyme assays are relatively easy and inexpensive to perform. A sample is taken (an individual) and enzyme tests are performed in the laboratory.

Another study on marine organisms (Narbonne et al. 1993) reports inhibition of ***acetylcholine-esterase*** activity in response to a pollution gradient. Metal contamination increased lipoperoxidation and PBC contamination induced the activity of detoxification enzymes and lipoperoxidation.

Activity of ***mixed function oxidase (MFO)*** enzymes is also related to detoxification. As with other enzyme indicators, the enzyme responds quickly to pollutants, within 1-3 days. This indicator has been used widely to detect impacts of pollutants on fish and more recently on mussels (Hansen 1995).

Hansen (1995) mentions several other indicators at the organism level, among those are indicators of the **immunological resistance** of organisms. Exposure to pollution may lead to immunosuppression, which can be measured by looking at the ***phagocytic activity***, i.e. the rate with which foreign particles and attached pollutants are digested. This has especially been used in mussels.

Another type of condition indicator is related to the **body reserves** accumulated in an organism. This indicator integrates information over a period and over a range of factors (including effects of pollutants and natural environmental variability). Since a change in the condition could have many causes, its indication is not very specific, but it may give valuable integrative and general information. In this way, it can show cause for alarm and suggest a more in-depth analysis of the problems.

Tissue weight relative to shell size has frequently been used as a condition indicator. This measure is given in the ***morphometric condition index*** (MCI) and is useful for an assessment of the ecophysiological or nutritional state of bivalves. One formula is MCI = shell-free dry weight / shell length^3 (Bonsdorff and Wenne 1989 for *Macoma balthica*). Others use the internal shell cavity capacity instead of shell length (Dame 1996). A decrease in MCI occurs during periods of environmental stress (pollutants, diseases, lack of food). This index is size dependent and different sites must thus be compared using individuals of the same size ("standard bivalves") (Bonsdorff and Wenne 1989). To interpret the index, it is therefore also preferable to know the population structure. It has been recommended that the molluscs be allowed time to cleanse the sediment from their guts before measuring these indices (Hawkins and Rowell 1987).

The ***energy content*** of bivalve tissue, measured using a bomb calorimeter, is another possible condition indicator (Crisp 1984). When exposed to stress, the somatic growth or reproductive effort of an individual is expected to change, and whether one or the other is affected, the nutritional condition of the individual decreases (corresponding to the condition of the individual) and thus the total energy content. Other condition indices are the glycogen and lipid content in the tissue or even the nitrogen content, but these are more laborious to obtain.

The ***humidity content*** (water content) of the tissue (all or specific parts, e.g. muscles or hepatopancreas) is simpler to obtain than the energy content and may provide similar information on the condition of an individual. The humidity level in muscles and liver has proven to be a good condition indicator in cod (Lambert and Dutil 1997), where the humidity increases when condition worsens.

A useful, non-destructive condition measure, which reveals the immediate potential for growth is the ***scope for growth*** (SFG). This needs to be performed in the laboratory. SFG is the energy available to an animal for gamete and somatic production after maintenance requirements are met. This can be expressed with the formula SFG = AB - (R + U), where AB is the absorbed energy, R is respiration, and U is the excretion (ammonia) (Bayne and Newell 1983). Scope for growth in bivalves has been noted to decline with increasing body burdens of toxic substances and to vary with varying seston quality (Capuzzo 1988). For SFG to be a useful tool in monitoring *in situ* pollution effects, the individual physiological components must be monitored within 24 hours of capture, i.e. before recovery can occur (Capuzzo 1988).

Another possible indicator using *M. arenaria* is the ***gonad glycogen content***. *M. arenaria* individuals that were transplanted into a polluted area (PBC, heavy metals [zinc and mercury] in Baie des Anglais) had decreased levels of glycogen and lipids in their gonads and increased fragility of the lysosomal membrane in the digestive gland (Pellerin-Massicotte et al. 1993). Pellerin-Massicotte *et al.* 1989 show that the gonad glycogen reserve is a good indicator for condition of an individual, since this reserve diminishes under adverse conditions. In addition, it is relatively easy to measure.

Another interesting indicator is **developmental stability** (Clarke 1995). This is the ability of an organism to withstand environmental and genetic disturbances encountered during development of a genetically determined phenotype. Under optimal conditions, development proceeds along a genetically determined pathway, and environmental disturbances affecting fitness-related traits are countered by stabilising mechanisms. Under too much stress, the stability mechanism is less efficient, resulting in differences in phenotypes that affect fitness. The morphological character to be used as an indicator has thus first of all to be correlated with fitness. If a character is not linked to fitness, a stress-related asymmetry does not reveal whether the individual is seriously affected by the stress. Secondly, the morphological character has to exhibit medium variability in relation to environmental variability. If it is too variable, too much noise will be present in the samples; if too stable, the indicator will not be sensitive enough.

A statistically significant positive correlation has been shown between the level of asymmetry in morphological characters of marine invertebrates and industrial pollution (Clarke 1993). In some cases, detectable changes occur before changes in direct fitness components (Clarke 1995). Developmental stability gives non-specific indications but is still very sensitive. It can be carried out non-destructively and is inexpensive and simple to perform. Clarke (1995) therefore recommends variation in developmental stability (e.g. fluctuating asymmetry) as a general early-warning indicator for environmental stress. The sample size needed to perform the analysis of a population is a minimum of 30 individuals (Clarke 1995).

The shape of the so-called scallops ears (lobes on the shells), *Chlamys islandica*, has been shown to be vaguely related to mortality, although with a positive relationship between survival and asymmetry (the size of ears of upper shell to ears of lower shell) (Fréchette et al. 2000). It remains to be seen whether the development of ear shape is linked to environmental conditions.

B. Growth

The relationship between growth, water flow and sediment disturbance is complex (Emerson 1990). Generally, *M. arenaria* seems to be relatively good at coping with high concentrations of suspended particles (compared to *Placopecten magellanicus*), even of poorer quality (Bacon et al. 1998, MacDonald et al. 1998). But although elevated particle concentration may lead to increased "scope for growth" due to increased food availability (MacDonald et al. 1998), a very high turbidity decreases growth in *M. arenaria* (Grant and Thorpe 1991). Any impacts affecting water flow and sediment disturbance could thus affect growth. In addition, the presence of toxic substances inhibits growth. Since growth varies with submersion time, using samples from the infralittoral area could simplify interpretation of growth measures.

The growth rate of an individual may be measured by **tissue growth**—measuring the change in dry tissue weight. The initial tissue dry weight of an individual can obviously not be determined directly (a destructive method). Instead, initial and final shell length and final dry tissue weight are measured and the change in dry tissue weight is subsequently estimated using a length-weight regression obtained from other individuals in the same population (Grant and Thorpe 1991). Under field conditions, such methods require the use of marked specimens.

The growth rate of an individual is often indirectly estimated by looking at ***shell growth***. Note, however, that shell growth is coupled to oxygen consumption, not necessarily to tissue production (Lewis and Cerrato 1997). Unlike tissue growth, shell growth is never negative in bivalves even under adverse conditions. This means that shell growth estimates may overestimate the tissue growth rate. So-called growth rings are produced under periods of suspended shell growth and the rings are often used to obtain age data that are crucial for estimating growth rate as well as recruitment and survivorship of a population. Due to the growth rings, the use of marked individuals is not needed to obtain the shell growth per year of an individual. These measures are taken under the assumption that one ring is created per year (during winter), although growth rings also may be produced in relation to other adverse environmental events (Kube et al. 1996).

A much more accurate measure for the soft tissue growth in bivalves is the ***RNA:DNA ratio***. This measure responds quickly to changes in the environment and is also used as an indicator for the growth potential (Mayrand et al. 1995). This can be measured for the tissue as a whole or for organs such as the digestive gland or the gills.

C. Reproductive capacity

One estimate of the reproductive potential of an individual is the **gonadosomatic index,** i.e. the gonad weight relative to the somatic tissue weight. This is also related to the condition of the individual (see above) and may be expressed in terms of changes in the condition index before and after spawning (Crisp 1984). Alternatively, it may be measured directly by measuring the output of females brought into the laboratory just before spawning.

Some important life history traits of *M. arenaria* have been shown to vary systematically with latitude. A potential indicator for effects of climate change (and associated changes in temperature) could perhaps be found among those traits. Mild winters were, for example, related to recruitment failure of *M. arenaria* in the Wadden Sea (Beukema 1992) and climate change could be expected to have similar effects. The life history traits concerned are life span, age at maturity, egg size and density, growth rate, and variation in juvenile mortality (Appeldoorn 1995). For example, the maximum age in Chesapeake Bay is reported to be 5 years, but over 20 years in Newfoundland. However, there are large unexplained site-specific variabilities in these traits, even at the same latitudes, which make them difficult to use in practice. Also, the age and size of *M. arenaria* at maturity in the relevant region is not known.

D. Behaviour

Although behaviour can be a powerful early-warning indicator under laboratory conditions, it is less applicable in the field. One possible indicator, however, is the ***burial depth*** of *M. arenaria*. Normal burial depth is correlated with the size of an individual. After being exposed during digging, individuals (1-7 cm length) living in mud reburied to shallower depths compared to undisturbed individuals (Emerson et al. 1990). Clams living in sand were much less affected because they were able to rebury faster and to attain their normal burial depth after exposure.

E. Disease and parasites

Disease decreases an individual's chance of survival and its ability to grow. Since an elevated **disease frequency** has been shown to correlate with pollution, it may serve as a useful indicator. External signs of disease can relatively easily be estimated on larger organisms, e.g. neoplasms (new or added tissue, generally pathological).

Pathological conditions have been used to create a ***severity index*** at the population level, using a matrix of occurrences and co-occurrences of pathological conditions (Lorda et al. 1981). Data on the incidence of five pathological conditions in a *M. arenaria* population (Saila et al.

1979 *in:* Lorda et al. 1981, Walker et al. 1981) were used to demonstrate the severity index (Lorda et al. 1981). The five pathological conditions used were neoplasia (new or added tissue), haemocytosis, hypoplasia (developmental deficiency), hyperplasia (overgrowth) and lipofuscin (a pigment within cells afflicted by atrophy).

Parasite load is another indicator that is related to the condition of the individual. Good condition allows better defence against parasites and a lower parasite load is thus expected. Diagnostic assays for detecting and quantifying parasites in *M. arenaria* have been developed for Atlantic conditions (McGladdery et al. 1993, McLaughlin and Faisal 1999).

5.3.2. Population level

A. Biomass

Biomass measures give coarse indications of the population development. It may be measured in dry or wet mass, with or without the shell (e.g. **kg/m^2**). *Mya* biomass on the North Shore of St. Lawrence varies in general between 0.14 and 1.5 kg/m^2, with a single location attaining 3.1 kg/m^2 (wet mass including shell). However, these data were gathered more than 25 years ago (Procéan 1995).

B. Productivity

Regular measures on *Mya* biomass may provide information on changes in productivity (e.g. **g/m^2/year**). Productivity, P, has been estimated using the equation: $P = \Sigma\ (N_t + N_{t+1})/2 \cdot \Delta\ w$, where N_t expresses the number of individuals at time t and w the weight (Crisp 1984, Günther 1992, Loo and Rosenberg 1996). Alternatively, the productivity can be determined using the age structure, taking into account the age–specific growth and mortality (and reproduction) (Cranford et al. 1985, Crisp 1984). As a shortcut to weight estimates, a length-weight relationship is sometimes used (determined from a few individuals). Using biomass for productivity estimates must be interpreted with care, however, since changes in *Mya* predation rates could confuse the picture.

Production to biomass (P/B) ratios may also be determined and used in comparisons. For example, P/B ratios for *M. arenaria* populations in Swedish waters have been calculated in the range 2.0 to 13.1 (Möller and Rosenberg 1983), whereas the P/B ratio for *M. arenaria* in a Nova Scotia inlet was found to be 2.5 (Burke and Mann 1974).

C. Abundance

Once initial abundance estimates have been made and objectives have been set, future estimates may be used as compliance indicators. The population density can be estimated by quadrate sampling, e.g. counting **individuals per unit area** (by digging) or siphon holes per unit area. Counting individuals by siphon holes is attractive because of its non-destructive nature, but seems to be difficult in practice since other organisms also dig holes. An indirect measure of density could be the ***catch per unit effort*** (C.P.U.). This is easiest to obtain for commercial harvesting; there is presently no surveillance on private harvesting of *M. arenaria*.

Variation in macrofauna abundance from deeper waters has been related to periodic (natural) variation in climate causing variation in nutrient availability (Hagberg and Tunberg 2000).

Any estimates of reference values should try to take such variation into account. As far as we know, there are no estimates of a minimal viable population size of *M. arenaria*.

D. Distribution (local)
The area or **number of square meters** that a *Mya* bed covers is an essential parameter, even just to determine the relevant study area. After initial determination of the size of a *Mya* bed, any changes over time can be used as a compliance indicator. Populations (or subpopulations) typically cover less than 1 km^2, but beds covering areas larger than 15 km^2 are also observed (Procéan 1995).

The distribution of a *Mya* bed is sensitive to changes in the physical environment. Kube (1996) found that during a prolonged stratification period, stocks located deeper than 10 m were strongly reduced by oxygen depletion. Thus, it is necessary to know the distribution in relation to water depths and tides.

Fractal dimensions of spatial distribution could possibly be used as a monitoring tool (Nelson 1996), using aerial photographs and GIS data, or simply measuring the spatial heterogeneity or complexity (Kent and Wong 1982, Seuront and Lagadeuc 1997).

E. Reproduction
Reproduction performance may be estimated by spawning output, larval densities, larvae settlement, or recruitment (spat). These parameters are highly variable and sensitive to a variety of factors. Most of them are quite laborious to obtain, thus restricting their use to diagnostic indicators, e.g. after other indicators have issued warnings.

The spawning itself is sensitive to temperature. A temperature threshold for spawning of about 10-12°C has been reported for both *M. arenaria* and *Macoma balthica* (references in Kube et al. 1996). After mild winters, a lower reproductive output has been observed (Kube 1996). Thus temperature changes may affect the spawning, and, among other things, the **spawning period**. Around the Manicouagan peninsula, the spawning period starts in mid May and continues to late June-mid July (Lamoureux 1977). Generally, there is a lack of a clear relationship between spawning output and recruitment success, rendering the prediction of the consequences of spawning variability difficult.

Larval density in the plankton was measured by Günther (1992) and related to settlement. Samples were taken using a net with 125 μm mesh size (Apstein net). Larval density by itself is highly variable in time and space and requires measurement at a specific time of the year. Therefore, it is less likely to be a good indicator of reproduction performance.

Although strongly influenced by local hydrography and sediment, *Mya* larvae preferentially settle where conditions for adult life are suitable. Therefore, **larvae settlement** (no. per unit area) generally indicates suitable habitat conditions for *M. arenaria*. Settlement densities are highly variable, e.g. from 2,800 individuals/m^2 to 13,000 individuals/m^2 within short distances (Günther 1992). High adult density may reduce settlement significantly (André and Rosenberg 1991). The measurement is quite labour intensive

Recruitment or the ***amount of spat*** (postlarval *Mya*) can be measured using core samples (number per unit area). Günther (1992) used mesh sizes of 125, 250, and 500 μm to obtain measures of this process and recorded around 5,000 postlarvae/m^2. Much larger numbers have been recorded (>> 100,000 postlarvae/m^2) (Günther, 1992). Recruitment success has been related to climate conditions, such as cold winter temperatures (Beukema 1992). Resuspension of postlarval *M. arenaria* has been observed due to currents or disturbance by other benthic organisms (Dunn et al. 1999). Recruitment integrates over all reproduction events from spawning to recruitment and is probably the most practical measure of reproduction.

F. Mortality

Mortality due to predation is sometimes estimated by exclosure experiments, but otherwise it is the age structure that is used to indirectly reveal mortality rates (see below) (Crisp 1984). Juvenile mortality is usually high in *Mya* populations due to predation by crabs, shrimp, and flatfish (Günther 1992, Kube 1996). Size-specific predation is due partly to selective feeding and partly to the larger *M. arenaria* being deeper in the sediment and thus better protected. Flatfish and shrimp have been observed to feed on clams up to a length of 17 mm (Kube 1996) and 3 mm (Möller and Rosenberg 1983), respectively. Another important predator is the nemertean *Cerebratulus lacteus* (Rowell 1992). In the St. Lawrence, *M. arenaria* is reported to be predated on by flatfish, crabs (*Cancer irroratus*), gulls, and snails (*Lunatia heros*) (Villemure and Lamoureux 1975).

G. Age structure

With samples of *Mya* specimens from a population, it is easy to obtain a **size–abundance curve**. In addition, the growth rings on the shells enable a translation of this into an **age–abundance curve** (Crisp 1984). Some authors suggest looking at internal growth lines instead of the external, which are less accurate (MacDonald and Thomas 1980). In both cases, accuracy decreases with age of the clams because the lines become difficult to distinguish. Tracking age structure over time may give detailed information on population dynamics because the survival of the different cohorts and the reproduction success can be inferred from following the curves from year to year (Günther 1992). Productivity may also be inferred (Burke and Mann 1974, Cranford et al. 1985).

A study in the southern Baltic Sea related the size and age structure of *Mya* populations to disturbance (waves, currents, erosion, and sedimentation), predation, climate variation, and oxygen depletion (Kube 1996). A high mortality rate during the first two years after settlement was apparently due to predation (size-selective predation affecting the abundance of young individuals). Rowell (1992) inferred from looking at the population structure that habitat change (in sediment and local hydrography) following construction of a tidal power plant rendered a site unsuitable for recruitment.

H. Genetic diversity

The level of heterozygosity measures genetic diversity. This may be measured at the molecular level, using DNA sequences as samples of the total genetic diversity (Burton 1996). Nuclear or mitochondrial DNA (or RNA) are used, and different methods exist for amplifying and sequencing the nucleotides. The segments are then separated by gel electrophoresis. Other estimates can be obtained by looking at different enzyme types (allozyme variation or allozyme polymorphism) using enzyme gel electrophoresis. This method is often cheaper and faster than the newer techniques based on DNA sequence variation (Burton 1996).

We know of no studies of the genetic diversity of *M. arenaria* population in the St. Lawrence, neither within nor between population diversity, whereas we have some information from the Atlantic coast (e.g. Morgan et al. 1978, Caporale et al. 1997).

5.3.3. Community level

A. Biomass

Biomass can be measured for each of the species present, for groups of species (e.g. Warwick 1986), or for whole communities. The units are usually mass per unit area (e.g. $\mathbf{g/m^2}$).

Schwinghamer (1981) reported biomass (volume) ranges for six intertidal communities in New Brunswick and Nova Scotia. The range varied from 70 cm^3/cm^2 to 1283 cm^3/cm^2 (the author used volume as a biomass measure). These numbers include bacteria, microalgae, meiofauna, and macrofauna communities. The macrofauna community biomass varied between approximately 5 and 1150 cm^3/cm^2, one of these (at 290 cm^3/cm^2) being dominated by *Mya arenaria*. The biomass measures were obtained for each of three size groups, micro-, meio-, and macro-organisms, using methods appropriate for each group (in order to obtain measures of size spectra, see below). Methods for measuring biomass of different communities can be found in Holme and McIntyre (1984).

B. Productivity

The biomass measures above furnish net productivity measures when tracked over time (e.g. $\mathbf{g/m^2/year}$). The ecological size spectra have been suggested as a means for surveying benthic productivity (see next section). As emphasised earlier, productivity estimated by tracking biomass might be underestimated due to other processes such as predation. In case of increasing production, there will be no measurable difference in biomass if the predation rate increases equivalently. Productivity measures based on population productivity considering both growth rates and reproduction are more precise, but also more laborious.

C. Diversity

One way of measuring diversity is **species richness**. Changes in this measure can give quite accurate assessments of stress related changes (Cairns et al. 1993). However, depending on the taxonomic groups involved, it may be difficult to obtain absolute measures of species richness because of taxonomic difficulties. Furthermore, the measure is sensitive to sample size, so that the larger the area examined or the more time spent looking for species, the more species are likely to be found. If species richness is used, it must be specified under which circumstances the measures were obtained (area, taxonomic detail, etc.).

The **Shannon-Wiener index** of species diversity is often used as a basic parameter for describing a community. The measure combines the number of species in a community and their relative abundance. Among the many diversity indices, it is probably the most widely used:

$H' = - \Sigma P_i*\ln P_i$, where i denotes the *i*th species in a community and P_i the proportion (in numbers or biomass) for the *i*th species ($i = \{1,\ldots, S\}$). The summation is performed over all species in the community.

Empirical evidence does not, however, suggest a consistent (unidirectional) relationship between diversity and environmental stress (Cairns et al. 1993 and references therein). This means that although the index can indicate change, it does not necessarily indicate if the change is an improvement or deterioration. Another point to note is that the index does not consider the identity of the species involved. Thus, changes in terms of species identity could occur without this being revealed in the index value. Generally speaking, the index should not be used in isolation.

Another related measure is species evenness, measured by equitability indices. For example, $J = H' / H_{max} = - \Sigma P_i*\ln P_i / \ln S$, where H_{max} is the maximum species diversity obtainable (i.e. when all species are equally frequent) and S is the species richness (i and P as above) (Begon et al. 1990). This index is less sensitive to sample size than the above index and is most often given along with H'.

As examples of the values that can be obtained, the following data are presented. Data from the St. Lawrence Estuary showed that in shallow waters (< 75 m), the number of mollusc species (including *M. arenaria*) was between 15 and 34 (Robert 1979). The same study recorded mollusc diversity (H') between 0.18 and 0.99. Diversity measures will obviously also depend on which groups are studied; for instance, a study on polychaetes from the St. Lawrence Estuary revealed species richness between 26 and 47 and diversity (H') in the range 0.85-1.22 (Massad and Brunel 1979).

A reduction in both species richness and diversity of marine benthos has been recorded in cases of pollution, e.g. severe oil pollution (Gray et al. 1990). But since there is not always a clear relationship between Shannon diversity and environmental stress, alternatives are sought. One alternative may be to use an increase in **between–sample variability of species diversity** (H') as a symptom of stress. Such an increase (compared to non-polluted sites) has been shown for macrobenthic replica samples stemming from an oil field in the North Sea (Warwick and Clarke 1993). The same study found that using multivariate analyses to reveal increased variability between samples was even more powerful (Warwick and Clarke 1993).

The **Warwick taxonomic index** (Warwick and Clarke 1995, Clarke and Warwick 1998b) is another interesting alternative. This taxonomic diversity index incorporates the average taxonomic “distance” between any two randomly chosen organisms from the sample. Whereas H’ treats all species similarly, the Warwick index takes into consideration that some species are more closely related than others and thus represent less diversity than remotely related species. There is some evidence that unperturbed benthic communities tend to contain more distinct species belonging to many different phyla than disturbed communities. In moderate and slightly disturbed communities, indices taking into consideration the taxonomic diversity and distinctness are especially valuable because they are more sensitive than the simpler diversity indices, such as H’ (Warwick and Clarke 1995).

The formula uses different weights (i.e. distinctness) for different taxonomic levels: species, genera, family, order, class, and phylum. The weights are defined as $w_1=1$ (species within the same genera), $w_2=2$ (species within the same family but different genera), and so on, up to $w_6=6$ (species in different phyla). Thus, very different species contribute relatively more to the diversity than do similar species. The taxonomic diversity (Δ) is defined as the average weighted path length between every pair of individuals (individuals of same species are weighted zero in the nominator and thus do not add to the diversity):

$\Delta = \Sigma\Sigma_{i<j}\, w_{ij}x_ix_j / (\Sigma\Sigma_{i<j}\, x_ix_j + \Sigma_i\, x_i(x_i-1)/2)$, where x_i denotes the abundance of the *i*'th species ($i = \{1,\ldots,S\}$) and w_{ij} is the distinctness weight for the path linking species i and j across the taxonomic tree (Warwick and Clarke 1995). The computation has now been included in some multivariate analyses software packages.

From this index of diversity (Δ), the **Warwick taxonomic distinctness**, Δ^*, is derived. This index tells the average weighted path length, when the paths between individuals of the same species are ignored in the denominator as well as in the nominator:

$\Delta^* = \Sigma\Sigma_{i<j}\, w_{ij}x_ix_j / \Sigma\Sigma_{i<j}\, x_ix_j$, where the denotation are as above (Warwick and Clarke 1995). This index, when applied to a marine macrobenthic community (at and near an oil field in the North Sea), showed a continuous decrease along a gradient of increasing environmental contamination (Warwick and Clarke 1995).

D. Dominance

Rank–abundance curves may be used to assess the responses of a community under stress. These curves show the relative abundance of each species on a log scale plotted against their species rank (Ramade 1995). Unpolluted areas should conform well to the so-called Preston's distribution ("broken stick," i.e. a relatively even distribution) whereas polluted conditions result in curves between Preston's model and log-linear ones (i.e. steeper curves). Polluted conditions favour the dominance of a few species, rendering the abundance curve steeper. This difference is equivalent to what is expected in early vs. late successional stages (Begon et al. 1990), where the unpolluted conditions relate to the later stages. These curves may be compared to the evenness index, J, since the evenness of a community also refers to the abundance distribution, although less detailed. The expectation also follows the trend predicted by Odum (1985), where disturbance tends to favour the dominance of r-selected species. This implies that any kind of disturbance may cause these changes, although we mostly find examples from pollution studies.

An example of a change in dominance in the community was given by Gilfillan et al. (1984). Following an experimental oil spill, infaunal communities changed when a number of opportunistic polychaete species bloomed. Their population density peaked three weeks after the spill and decreased as the hydrocarbon level in the sediment declined, after approximately 9 weeks (Gilfillan et al. 1984).

In **K-dominance curves,** the species are ranked in order of importance (in terms of biomass or abundance) on the x-axis, just like the rank–abundance curves, whereas their percentage dominance is plotted on a cumulative scale on the y-axis (Lambshead et al. 1983). We expect that unpolluted communities will become increasingly dominated by a few, large K-species (i.e. species with attributes of large body size, long life-span, and a fairly constant population size that is close the carrying capacity). In unpolluted marine benthic communities, these K-species will dominate in terms of biomass but not abundance. In contrast, the dominance of these species will decrease in disturbed communities, thus leading to a steeper K-dominance curve.

To detect pollution effects, Warwick (1986) suggested the drawing of two K-dominance curves (or ABC for abundance/biomass curves), one ranking the species in terms of biomass and one in terms of abundance. In benthic communities close to some equilibrium, the dominating species will dominate in biomass but not in abundance (Fig. 10). Thus, a “healthy” community would exhibit a less steep logarithmic curve for the biomass ranking but a more steeply increasing logarithmic curve for the abundance. Under polluted conditions, the curves will shift, so the biomass curve will become more steep and the abundance curve less steep as the conditions worsen (Warwick 1986). This model allows an assessment with a fixed benchmark, i.e. where the switch of dominance type (biomass or abundance) occurs.

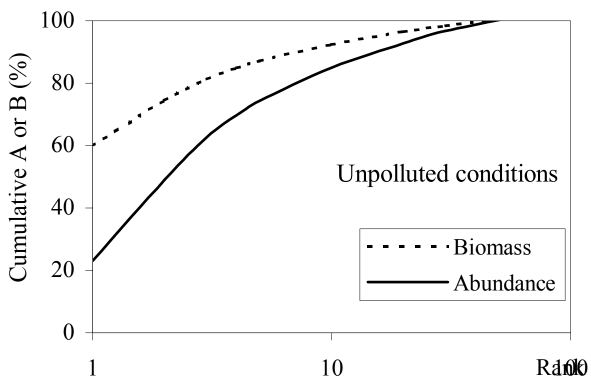

Fig. 10. Hypothetical K-dominance curves for species biomass (B) and abundance (A) under unpolluted conditions. Under polluted conditions, the slope of the biomass curve will become steeper whereas the abundance curve will become less steep, i.e. the curves will switch places (redrawn from Warwick 1986).

K-dominance curves (ranking of both species abundance and biomass) have been drawn for polluted and unpolluted macrobenthic communities, showing that these may be used to detect the pollution effects on a community (Warwick 1986, Warwick et al. 1987). Further development of this methodology has taken place (Clarke 1990, Warwick and Clarke 1994).

The use of the nematode:copepod ratio (Raffaelli and Mason 1981) as an indicator is based on the assumption that since copepods are more sensitive to pollution than nematodes, an effect of pollution would be reflected in the ratio of their abundances. However, this ratio has proved to be too unpredictable (Gee et al. 1985).

E. Species composition

Both natural and anthropogenic disturbances may cause defaunation of marine sediments. Changes in community composition along pollution gradients have been characterised through changes in species number, abundance, and biomass (Pearson and Rosenberg 1978, Newell et al. 1998). An experimental study on recolonisation and succession of marine macrobenthos (comprising polychaetes, gastropods, and bivalves and others) showed that temporal changes in abundance, species number, and diversity following an experimental defaunation of small areas (approx. 0.08 m^2) were similar to the changes along a decreasing pollution gradient (Lu and Wu 2000). The species composition changed significantly over time to more or less complete succession within 15 months. In temperate regions, this recovery may take several years (Boesch and Rosenberg 1981). Here, r-strategists (opportunists with life-history traits such as small size, fast growth, and fast reproduction) are usually also the first colonisers, whereas a shift toward more competitive species occurs later in the succession (e.g. Pearson and Rosenberg 1978, but see Zajac and Whitlatch 1982a, 1982b).

Many studies have shown effects of thermal pollution on benthic communities (references *in* Lardicci et al. 1999). However, the level of impacts ranges from none to large, depending on local conditions. One study of the soft bottom community that took into account natural variability did not find any additional changes at the polluted site (1-2°C more than ambient water) (Lardicci et al. 1999). Evidence suggests that the effect of thermal pollution on species composition should not be sought at the shallower depths, where the natural variability and thus the tolerance for changing temperature is higher. The same would hold true for effects of climate change.

Multivariate analyses, i.e. classification and ordination (multidimensional scaling), are able to detect effects of oil pollution of lower intensity than diversity measures (Gray et al. 1990). The first changes are increased abundance of some species and changes in presence–absence patterns of rare species, whereas the opportunistic species only dominate under severe pollution. Multivariate analyses have also been used to reveal increased variability among replicate samples associated with pollution (Warwick and Clarke 1993). Here, a **comparative index of multivariate dispersion** (IMD) was suggested to measure the variability. The index compares the rank order of dissimilarities between samples, the rank orders being contained in the dissimilarity matrix underlying the multidimensional scaling. More precisely, having re-ranked the matrix ignoring all between-treatment dissimilarities, the average rank of dissimilarity among impacted samples (r_t) are contrasted with the average rank of dissimilarity among control samples (r_c):

$IMD = 2(r_c - r_t)/(N_c + N_t)$, where $N_c = n_c(n_c - 1)/2$, $N_t = n_t(n_t - 1)/2$ and n_c and n_t are the number of samples in the control group and impacted group, respectively. IMD has maximum value of +1 when all dissimilarities among impacted samples are higher than any dissimilarities among control samples and a minimum value of -1 in the converse case. Values near zero indicate no difference between treatment and control groups.

The effect of trace metals on infauna composition has also been determined by using classification and ordination (Ward and Hutchings 1996). In addition, the contribution of trace metals in structuring macroinvertebrate composition has been estimated by variation partitioning (partial canonical correspondence analysis) (Peeters et al. 2000).

Finally, invasion of exotic species has been shown to be facilitated in disturbed environment. This occurrence may also be surveyed by looking at species composition (e.g. Currie and Parry 1999).

Clarke and Warwick (1998a) quantified the **structural redundancy** within marine macrobenthic communities by looking at community responses to oil spill or natural variation. The redundancy was expressed by the number of subsets of species that produced similar response patterns as the total data set (shown by multivariate analyses, viz. multidimensional scaling). This means that each subset of species covers approximately the same environmental variation as the full data set. When this can be related to functional aspects, the redundancy may serve as an indirect measure of the resilience of a system (i.e. the structural or taxonomic redundancy replicates the major functional groupings).

5.3.4. Ecosystem level—biological component

A. Trophic biomass and productivity

The primary production is an important feature of the productivity of an ecosystem, since it lays the foundation for the production at higher trophic levels. A range of methods exist for measuring primary production (Therriault et al. 1990, Bourgeois 1996). One is measurement of **oxygen production** during an incubation period; another is measurement of **carbon-14** (radioactive carbon) incorporation. Other methods use **stable isotopes** or measure **metabolites** associated with production. In some cases phytoplankton biomass can represent the intensity of production. Phytoplankton biomass can be estimated by quantifying **chlorophyll pigments** or **ATP**. Nieke et al. (1997) used a ship-borne laser fluorosensor to **remote sense chlorophyll *a*** concentrations in a coastal environment in the St. Lawrence Estuary and thereby estimate phytoplankton biomass. Measurements obtained this way were accurate estimates of surface chlorophyll *a* concentrations extracted from water samples. These estimates can be obtained from low-altitude flights, preferably performed at dawn.

The biomass and productivity of the higher trophic groups, i.e. herbivores and carnivores (usually there are three or four trophic levels), may also be determined. In the absence of direct productivity measures (that are costly and laborious), the **zooplankton biomass** may be used as an indicator of secondary production in the water column (Bourgeois 1996), more specifically, the copepod biomass. This is not a very good measure, however, since copepods reproduce continuously and the biomass varies considerably over time and space. Rather, indirect measures of productivity should be used. Since copepod adults no longer contribute to production by growth, the estimates may be based exclusively on egg production and larval recruitment, measuring the **biomass (or number) of females, their fecundity, and the hatching success of the eggs** (Poulet et al. 1995, Runge and Roff 2000). These variables are sensitive to disturbance and pollution (e.g. Cowles and Remillard 1983).

The tertiary production may be estimated from biomass estimates (plus reproduction estimates), or catch per unit effort measures obtained from fisheries. For the top trophic group, there are other measures to be considered (see “top predators” below).

If biomass measures for both autotrophs and heterotrophs are available, an indicator of resilience can be obtained. This is the **heterotroph:autotroph biomass ratio**, where a high ratio should indicate high resilience (Begon et al. 1990). This is related to rate of flux through the system. A high ratio reflects a short life and rapid turnover of the phytoplankton, and thus an ability to "flush" a perturbation through the system. This holds true for both energy flow and nutrient cycling (Begon et al. 1990, DeAngelis 1980) when comparing different types of ecosystems. Perhaps the ratio is also of value for within-ecosystem comparisons. A similar ratio may be obtained (for relatively open waters) by measuring chlorophyll *a* (approximates phytoplankton biomass) and particulate protein (estimate of total biomass), i.e. the **chlorophyll:protein ratio** (Dortch and Packard 1989).

A trophic state index has been used in relation to eutrophication in lakes. This is based on measurements of chlorophyll *a*, water clarity, and total nitrogen and phosphorus in the water column (Hunsaker and Carpenter 1990 *in*: Cairns et al. 1993). This measure also integrates the amount of primary consumers. We do not know whether such an index could be useful for problems other than eutrophication or for the ocean. Another **trophic state index (O_2)** has been developed for estuaries and is based on oxygen concentrations in the surface and bottom waters. The index is the Euclidean distance between the two (transformed) oxygen concentrations, both relative to the oxygen content at 100% saturation (see Justić 1991).

The **ecological size spectra** (Schwinghamer 1981) are described as characteristic size distributions of benthic communities. A size spectrum comprises bacteria/microalgae, meiofauna, and macrofauna. Characteristic peaks were found in graphs depicting biomass on a log scale (measured by volume) in relation to the diameter of the organisms on a log scale, such graphs being called Sheldon spectra (from Sheldon et al. 1972). The diameter of non-spherical organisms is calculated as the diameter of a sphere of equal volume (the "equivalent spherical diameter"). The typical features of the Sheldon spectra for benthic communities are biomass maxima at the lower and upper extremes of the size scale plus a local maximum at the centre of the graph.

The ecological size spectra have been applied to different benthic communities, where the characteristic distributions reappeared (Schwinghamer 1981). Therefore, they may be useful for comparing communities and detecting environmental disturbances. Schwinghamer (1981) thus suggested that deviations from a typical spectrum can be used to detect exogenous disturbance of ecosystems, e.g. pollution, or to survey benthic productivity. If these effects could be surveyed by biomass-size spectra alone, time consuming taxonomic studies may be avoided at least as a first step (Schwinghamer 1988). Kerr and Dickie (1984) suggest that deviations from typical patterns should be used in general as indications of ecosystem health problems. Evidence partly supports these suggestions (Schwinghamer 1988), but the method has not yet been widely used and practical applications are generally lacking.

B. Trophic diversity

Determining this type of diversity means categorising each of the considered species according to their trophic level. It is not always clear which trophic level a species belongs to, since the same species may feed on prey from several levels. Making a weighted average for the trophic levels of the prey species (TLj) and adding one may solve this:

$TL_i = 1 + \Sigma (DC_{ij} * TL_j)$, where i is the predator, j is the nth prey and DC_{ij} is the diet composition expressing the fractions of each j in the diet of i. The summation is done over all prey species ($j = \{1,\ldots, n\}$) (Christensen and Pauly 1992).

Alternatively, if it is advantageous to work with trophic levels only in integers, each predator may be "split" according to the prey proportions consumed from each level (Christensen and Pauly 1992). Trophic diversity may simply be the **number of trophic levels** in a system, but a **trophic diversity index** analogous to the Shannon-Wiener index (see community level) has also been suggested (Brown et al. 2000).

C. Trophic structure or composition

The trophic structure or composition can be considered for the biological component of the ecosystem as a whole or for a single community. Here, the parameter is placed under the ecosystem level, although the indicators mentioned in the following section are often applied to the benthic community level. We place it here because trophic levels and functional groups, such as carnivores and suspension feeders, are used in the indices. Functional groups traditionally pertain to ecosystem ecology.

Recently, a **trophic structure index** has been proposed (Done and Reichelt 1998). This index uses a pyramid structure of the trophic levels, but with each trophic level subdivided in two, an adult and a juvenile portion. Each of these levels are given a weighting, j, where j = {0.5, 1, 1.5,…., 3} when there are six levels. The value of the trophic structure index is calculated thus:

$V_{ts} = \Sigma (c_j * \alpha^{j-1})$, where c is the proportion of biomass for level j, α is proposed to be set at 10, thus weighting each trophic level ten times the one below (refering to the 10% rule of thumb for ecological efficiency of energy transfer between levels). The summation is done over all levels. This index does not seem to have been applied yet.

Another recently suggested index is the **mean trophic level**. It has been used as an index of sustainability for fisheries, calculating the mean trophic level of landed fish (Pauly et al. 2001).

The trophic structure of macrobenthos has been shown to change from a dominance of suspension feeders to deposit feeders along a gradient of increasing organic pollution (also in relation to oil pollution) (Pearson and Rosenberg 1978). A change in the **ratio of suspension to deposit feeders** in the sediment may be used as an indicator of increasing or decreasing pollution (the percentage of deposit feeders increases towards a maximum with increasing organic input).

The **benthic index of biotic integrity** (B-IBI) was developed for monitoring the state of Chesapeake Bay (Weisberg et al. 1997) following the principles of Karr et al. (1986). The measures, or so-called metrics, were developed based on the paradigm that benthic assemblages respond to improved habitat quality by increasing in abundance, increasing in species diversity, and a change in assemblage from pollution-tolerant to pollution-sensitive species (Pearson and Rosenberg 1978). Weisberg et al. (1997) add that abundance and diversity deeper in the sediments should increase and that feeding guild diversity should increase. The index succeeded in distinguishing between polluted and unpolluted sites 93% of the time. For the most saline habitats, the metrics of the benthos were: Shannon-Wiener index, abundance (no./m^2), biomass (g/m^2), pollution-tolerant taxa (% biomass), pollution-sensitive taxa (% abundance or biomass), deep deposit feeders (% abundance), carnivores and omnivores (% abundance), and taxa below the sediment–water interface (%). The data collection would thus require biomass measures and depth-specific sampling, both of which are time consuming. However, dropping the biomass and depth-specific metrics result in only a minor reduction in classification efficiency (from 93% to 92%).

A similar index would probably be successful in assessing the St. Lawrence coastal areas, although the problems may be of a different nature and an adjustment to local issues is needed (lower levels of pollution and other types of impacts). In addition, calibration of the index with local conditions and local species composition is required. A new index should preferably be calibrated with multiple control sites.

A measure of connectivity in food webs has also been used to describe the characteristics of an ecosystem. The **connectance** for a given food web is the ratio of the number of actual links to the number of possible links in a food web (Christensen 1995). However, the relationship between ecosystem development and connectivity is not clear. Some find that critical values of connectance are related to stability, others that stability decreases with increasing connectivity (Christensen 1995). In addition, connectance depends to a large extent on how a system is described, thus making it difficult to compare between systems. Within-system comparisons may nevertheless prove useful.

D. Top predators

The **abundance and reproductive parameters of top predators** seem likely to be indicators of ecosystem changes. Top predators bioaccumulate contaminants, making it easier to detect the chemicals before they have an effect on other organisms. This makes them vulnerable to morphological, behavioural, and physiological effects that are easy to observe, i.e. **sublethal effects** at the level of the individual. Birds and mammals are particularly good indicator candidates (Burger and Peakall 1995) because their biology is well known compared to other organisms and baseline data are therefore more often available. These animals are also so visible to the public that changes in their health and population levels are likely to be noted and cared about.

Effects on population parameters have been demonstrated. For example, some marine seabirds failed to reproduce as a response to lack of food resources, thus exhibiting density-dependent responses to food availability (Bost and Le Maho 1993). Bost and Le Maho (1993) believe that sea bird breeding parameters show promise for future use, but that food abundance and foraging efficiency need to be measured in concert to get earlier warning and a stronger link to any failures in breeding success. Birds have also been used as monitors of oil pollution and other pollutants (Furness and Camphuysen 1997). Especially, birds bioaccumulate heavy metals and chlorinated contaminants (St. Lawrence Centre 1996, Furness and Camphuysen 1997), as shown by eggshell thinning. These effects give a good measure of the level of contaminants at the ecosystem level.

Herring gulls have been intensely studied with respect to the impact of pesticides on the Great Lakes (Cairns et al. 1993). In addition, the population dynamics of herring gulls on the North Shore of the Gulf of St. Lawrence seem to be related to cod fishery activities (Chapdelaine and Rail 1997), thus showing a link to the top of the marine food web. The predator is present year round and widely distributed, all of this pointing to the bird as a good potential indicator. A downside to using this bird as an indicator is its relatively broad tolerance for various environmental factors. However, if various benchmark levels can be set, this is not a hindrance for its use.

Marine mammals are particularly sensitive indicators of a wide variety of organic and inorganic toxic substances (St. Lawrence Centre 1996) that accumulate in the kidneys, liver and muscles. The beluga could serve as an indicator species due to its geographic position and its place at the top of the food chain. The long life span of the beluga, however, means that it is quite slow at detecting environmental changes. On the other hand, the long life span makes the beluga more susceptible to bioaccumulation of various contaminants than other organisms. Since we are working in the littoral zone, it may be more relevant to concentrate on seals. The harbour seal might serve as an indicator species, being faster to reveal effects than the beluga, due to shorter life-span.

5.3.5. Ecosystem level—chemical component

A. Nutrients

Nutrient measures from the sediment or water column (Bryan et al. 1985) may best serve as diagnostic indicators when interpreting biological indicators. The same applies to oxygen concentrations. Measurements of nitrogen and phosphorous contents in various forms and fluxes between the sediment and water column may be conducted *in situ* along with oxygen measurements (Hopkinson and Wetzel 1982).

B. Nutrient cycling

The **depth of the blackened sediment zone**, indicating the start of anoxic sediments (Flint and Kalke 1983), may be used to track changes in the environment. Areas treated with chemical poison (methyl parathion) exhibit this layer 1-2 cm beneath the sediment surface while untreated areas showed the start of the zone deeper than 3 cm (Flint and Kalke 1983).

The increased use of organic contaminants means adding organic material to the sediment. This in turn leads to an initial increased biological oxygen demand (BOD) in the sediment when the organic matter is decomposing, both from the toxin itself and the increased number of dead animals (Flint and Kalke 1983). This means that ammonium regeneration rates are higher during this time period. After some time (80 days in Flint and Kalke 1983), a change in the infaunal community structure leads to decreased ammonia fluxes, lower metabolic rates (sediment oxygen uptake), and a shallower oxygenated zone. A shallower oxygenated zone is also seen following other organic pollution, i.e. eutrophication (Pearson and Rosenberg 1978, Newell et al. 1998). Changes can also be expected following sediment disturbance.

The **Finn cycling index** (Finn 1980, Christensen 1995) has been used to describe nutrient (and energy) cycling within a system. The index is calculated as the proportion of nutrient (or energy) flow that is recycled in the system. Calculating this index demands a good knowledge of the system and its components. The related **predatory cycling index** (Christensen 1995) has the same drawback. Another index is the **average path length of an element**, which is the average number of system units (boxes) that an inflow or outflow element passes through (Finn 1980). This is affected by the number of trophic levels, amount of cycling, system complexity, and the degree of detail used in the model (model resolution). The average path length is generally expected to be highest in more mature systems (Christensen 1995).

C. Contaminants

Direct measures of contaminants can be used as diagnostic indicators when related to biological indicators, e.g. measures of the **sediment content** of contaminants. The contaminants may include heavy metals, chlorinated organic compounds, and polycyclic aromatic hydrocarbons. These direct measures are, as earlier mentioned, pressure indicators rather than state indicators, since the concentrations do not reveal the effects on the organisms. In general, toxicity measures are most relevant when a contaminant or range of contaminants is pointed out by a given major issue; otherwise, there are simply too many chemicals to consider.

An indirect measure is the **tissue content** (concentration in the flesh) of a particular stressor. For example, the concentration of lead in the shell of *M. arenaria* is correlated with dissolved lead in seawater, but concentrated by a factor 10,000 (Pitts and Wallace 1994). Threshold levels for human consumption of *Mya* have been set for some heavy metals (arsenic [3.5 μg/g], mercury [0.5 μg/g] and lead [0.5 μg/g]) and for some organic contaminants (Biorex 1999c). Not all organisms are equally suited as indicators because of their different habitat preferences and different absorption rates. The suitability of marine benthic organisms as indicators of metal contamination is relatively well examined, so that depending on the metal of interest, a certain sensitive species can be chosen (Bryan et al. 1985). *Macoma balthica*, for example, is considered as a good indicator of mercury pollution whereas *Nereis versicolor* is more sensitive to copper (Bryan et al. 1985).

In addition, *Mya arenaria* has been used as a bio-indicator for the anti-fouling poison used for boats, TBT (Kure and Depledge 1994). It was concluded that *M. arenaria* shows potential as a bio-indicator of TBT pollution because it seems to have very limited ability to metabolise and eliminate TBT but at the same time tolerate relatively high concentrations (Kure and Depledge 1994).

The blue mussel, *Mytilus edulis*, can serve as an indicator of, for example, petroleum pollution in coastal waters (Burns and Smith 1981) since it accumulates hydrocarbons. Body burdens in bivalves can thus be used to monitor levels of chronic oil pollution. *M. edulis* is also used as an indicator of metal pollution (Meeus-Verdinne et al. 1984, Gibb et al. 1996).

5.3.6. Ecosystem level—physical component

The parameters measured at this level are often measured as background values and thus they serve most often for diagnostic purposes when biological indicators are examined in detail. The landscape (or bottomscape) and biogenic structures should be considered in this section, but we do not at this point attempt to identify indicators for these structures as they depend more heavily on local conditions.

A. Sediment/substrate

Increased erosion or sedimentation that could change the composition of the sediment are measured by a **change in the granulometry** of the sediment. Since benthic organisms have different preferences, this may change the community composition considerably. For example, one study noted a change in sediment composition following the installation of a tidal power generating facility, which prevented the settlement of larval clams (Rowell 1992). Granulometry can be performed on a block of sediment (mixed) or, for a profile, by measuring the granulometry on each of a number of layers. The latter indicator is more time consuming, but also more informative and sensitive.

Increased sewage influx leads to an increase in the **percentage of organic matter** in the sediment. Changes in hydrography could also result in changes in this respect. Change in organic content may also profoundly alter the oxygen regime in a benthic community (e.g. Pearson and Rosenberg 1978).

B. Currents

Change in currents (direction and strength) is mostly seen by changes in sedimentation rate and thereby changes in sediment composition, i.e. the indicators mentioned in the above paragraph.

C. Turbidity

An important parameter in the physical compartment is turbidity. Turbidity increases as a result of an increased input of sewage or nutrients, or an increased disturbance of the sediments. It is measured in so-called **nephelometric turbidity units** (St. Lawrence Centre 1996), which are determined by measurements of light scattering. The more particles in the water, the more the light is scattered and the higher the turbidity. This affects the light penetration depth and thus the potential for primary production. An alternative method is to measure the **Secchi depth**, i.e. the depth at which a white and black dish can no longer be seen in the water column (from the surface). As mentioned earlier, high turbidity may also affect bivalve feeding rates.

Additional parameters

There are several important environmental variables that need to be measured in order for us to be able to explain natural variation. These are air temperature, water temperature (although highly variable in the intertidal zone), salinity, turbidity, storm occurrences, granulometry, currents, and immersion time (water depths). Some of these parameters can probably be acquired (with some adjustments) from existing monitoring programs (Vandermeulen 1998).

5.4. Evaluation and selection of indicators

In the preceding section, we listed indicators relevant for the major issues regarding the *Mya–Macoma* community. In the text, the indicators are ordered by the organisational levels of the ecosystem, while the indicators pertaining to each major issue (fisheries, pollution, habitat modifications, and climate change) can be seen in Fig. 6-9.

In order to proceed with building a monitoring program, we need to select a set of indicators among the possible indicators. Since monitoring is repetitive and resources are limited, we need inexpensive indicators and the smallest number of indicators that still provide the desired information. These demands (and compromises) are reflected in the criteria that indicators should fulfil (listed in Table 2).

Evaluating the indicators according to the criteria in Table 2 is a way to select a set of indicators, which is what we have done in the following section. It should be noted that the most important criteria are not the same for all types of indicators. For example, it is important for an early-warning indicator to be timely and less important that it be integrative compared to compliance indicators. In addition, no indicator is perfect and most indicators are associated with methodological problems. The selection process should therefore seek to correct these flaws by selecting a suite of indicators that together cover the essential criteria, thus enabling a program to shed sufficient light on the ecosystem of interest.

As part of the selection process, it is also relevant to restate the objectives and to go into more detail with them. The specific objectives must in each case be clearly stated for the biological community and the area in question. The objectives may be stated explicitly, using the properties and organisational hierarchy presented earlier in the report. An example of an objective for a given zone could be that the *Mya* population should not decrease to less than 85% of its initial (historic) population size. Focusing on objectives helps to keep the purpose of the monitoring program in mind, i.e. what it is that we want the indicators to tell us.

5.4.1. Selection of a set of indicators

In the following section, we select a set of indicators for the *Mya–Macoma* community. These indicators are selected following a review of all indicators presented for each of the four major issues (see Fig. 6-9). Each indicator is evaluated according to the indicator criteria (Table 2) with respect to a certain issue. The chosen set of indicators is shown in Table 3.

A. State indicators

Individual level indicators
At the individual level, condition, growth, and behaviour are the properties for the fisheries issue whereas the properties for pollutants are condition and disease/parasite frequency of the organisms. For habitat modifications, condition and growth are the most important properties while the condition and reproduction properties are especially expected to react under climate change.

Table 3. The indicators proposed for each of the major issues concerning the *Mya–Macoma* community. These indicators were chosen following a review of all the indicators presented for each of the four major issues at the various ecosystem levels (Fig. 6-9). The letters in the last columns indicate which issue(s) the indicator pertains to: F = fisheries, P = pollution, H = habitat modifications, C = climate change.

Properties	Indicators	F	P	H	C
Biological ecosystem component					
Individual level					
Condition	Enzyme activity		x		
	Morphometric condition index	x		x	x
	Energy content	x		x	x
	Gonad glycogen content	x	x	x	x
Growth	Shell size	x		x	
Reproductive capacity	Gonadosomatic index				x
Behaviour	Burial depth	x			
Disease/parasites	Disease frequency		x		
Population level					
Biomass	g/m^2	x	x		
Productivity	g/m^2/year	x	x		
Abundance	Density (no./m^2)	x	x	x	
	C.P.U.	x			
Distribution	Area (m^2)	x		x	x
Reproduction	Recruitment (no. spat/m^2)		x	x	x
	Spawning period				x
Age structure	Size structure	x			
Community level					
Biomass	g/m^2	x	x	x	x
Productivity	g/m^2/year	x	x	x	x
Diversity	Species richness	x	x	x	x
	Shannon-Wiener index	x	x	x	x
	Warwick taxonomic index	x	x	x	x
Dominance	K-dominance curves	x	x	x	x
Species composition	(Multivariate statistics) IMD	x	x	x	x
Ecosystem level					
Trophic biomass/prod.	Ecological size spectra	x		x	x
Trophic struct./comp.	B-IBI	x	x	x	x
Top predators	Reproductive success		x		
	Sublethal effects of contaminants		x		
Chemical ecosystem component					
Nutrient cycling	Depth of black layer	x	x	x	x
Contaminants	Tissue content of contaminant		x		
Physical ecosystem component					
Sediment/substrate	Granulometry	x		x	x
	% organic matter	x	x	x	x
Turbidity	Nephelometric turbidity units		x		

Of the many condition indices listed, we suggest to focus on the ***morphometric condition index** (MCI)* and ***energy content*** for three of the major issues, all except pollution. The MCI is scientifically valid and integrates the conditions faced by the bivalves. It is also cost effective and easy to measure compared to the other possible indicators. On the other hand, energy content is a more precise measure and is expected to complement the MCI. ***Glycogen content in gonads*** is believed to be useful for all four issues. Gonads glycogen content is more difficult to measure than the total energy content, but it has already proven to be a valuable indicator for the condition of *Mya* individuals. As an index suited for the pollution issue, the ***enzyme activity*** is suggested; more precisely, a general enzyme such as glucose 6-phosphate dehydrogenase. That it responds to pollution is well documented, it is a sensitive and timely indicator, and it is not too complicated to measure.

As an indicator of growth, which is an important property under fisheries and habitat modifications, we suggest **s*hell growth***, since it is much simpler to deal with than tissue growth. This is done in spite of the risks of error related to using the shell lines.

A reproduction indicator is the ***gonadosomatic index***. This is an indicator of reproductive potential. It is scientifically valid and relatively easy to measure, although limited to measures performed at a specific time of the year.

As an indicator of the behaviour of *M. arenaria* a recording of the ***burial depth*** (in relation to shell size) is suggested. This is most relevant for fisheries (digging, sediment disturbance), and it is a timely measure relatively easy to obtain.

Finally, an index suited for the pollution issue is the disease/parasite frequency, notably the ***severity index***. The scientific validity of the indicator is well established for pollution. In addition, it is integrative and can possibly be tied to standards, i.e. benchmarks. The disease parameter is suggested to be used for pollution because this indicator allows for higher accuracy and specificity in detecting problems.

Population level indicators

At the population level, we seek indicators for the biomass, productivity, abundance, distribution, and age structure of *M. arenaria* populations in relation to the fisheries issue. For the pollution issue, the properties to be monitored are biomass, productivity, abundance, and reproduction. For habitat modifications, abundance, distribution, and reproduction are the main properties affected. For climate change, distribution, and reproduction properties are probably the most sensitive ones.

Biomass measures are obtained from population samples, measuring dry weight or ash free dry weight (***g/m^2***). We suggest that productivity (***$g/m^2/year$***) be estimated using the formula $P = \Sigma\ (N_t + N_{t+1})/2 \cdot \Delta\ w$ (section 5.4.2. C). This does not take into account the age structure and the reproduction, which makes it relatively easy to obtain.

For abundance and distribution, we suggest that the properties also serve as indicators (***$no./m^2$*** and ***m^2***). It is possible that a ***catch per unit effort*** measure can be obtained for the fishing, but we suggest to also use the more accurate estimate of abundance.

The suggested indicator of age structure is ***size structure***. Measuring the annual rings to estimate age would increase accuracy but decrease cost effectiveness and ease of measurement. Size structure has been used frequently and is a quickly obtained and integrative indicator. An improved measure is obtained from counting the annual rings on a subset of the sampled individuals to obtain a population-specific translation of size into age. This represents a valid compromise where increased accuracy is joined with cost efficiency.

As indicator of reproduction, we suggest the recruitment rate (***no. spat/m^2***). This reproduction measure is easier to obtain and a better integrator than measuring, e.g. larval settlement. Another indicator, which is useful under the climate change issue, is the ***spawning period***. Spawning period is directly related to temperature, thus sensitive to a change in temperature.

Community level indicators
At the community level, we have to look at the total biomass and productivity as well as the entire species composition for all four major issues.

Total biomass and community productivity are obtained by following the biomass and the change in biomass over time, measured in dry weight or ash free dry weight (***g/m^2*** and ***g/m^2/year***). These are integrative measures are fairly rough but simple to report and meaningful to communities.

Species composition is examined using ***multivariate statistics*** (*IMD*). If an effect is detected in the species composition and the effect is found to be consistent, it is possible that diversity or dominance measures could suffice. Suggested indicators of these properties are the ***species richness***, the ***Shannon-Wiener index***, the ***Warwick taxonomic index*** and ***K-dominance curves***. All of these indicators are scientifically valid, but the first two are not very sensitive to medium and low intensity disturbances. However, they have been widely used and their reactions under various circumstances are relatively well known.

Biological ecosystem component indicators
At this level, the productivity is chosen for all issues but pollution, where top predator indicators are chosen instead. For all four issues, the trophic composition and structure are relevant. At present, we wish to avoid monitoring at this level until an effect has been clearly documented at the community level. However, the information on the trophic structure and composition can also be used at the community level, i.e. where only organisms within the benthic community are examined with respect to trophic levels.

To obtain estimates of productivity, ***ecological size spectra*** measuring biomass of several communities in the sediment are suggested as indicators. This indicator roughly follows the distribution of biomass among different communities and thus gives information on productivity in an integrative manner.

The indicator of trophic composition or structure could be similar to the ***B-IBI***, which then needs to be adapted to local conditions.

For the pollution issue, the effect of the accumulation of contaminants in top predators and their reproduction are important to follow. Suggested indicators are the ***reproductive success*** and ***sublethal effects of contaminants*** in piscivorous or benthivorous birds. Birds are relatively well studied, thus giving a good basis for scientifically valid indicators. The information is laborious but feasible to obtain, and it might be obtained from other programs.

Chemical ecosystem component indicators
In the chemical component of the ecosystem, all four major issues could cause interference with the nutrient cycling.

During a significant change in the nutrient cycling, the ***depth of the black layer*** in the sediment is expected to change (i.e. the depth of the oxygenated layer changes). This is an integrative measure and as such does not furnish specific information on nutrient cycling. It is unavoidable that some cycling changes pass undetected using this rough measure. On the other hand, it is fast and very easy to measure, as opposed to any direct measures of nutrient cycling.

For the pollution issue, the content of contaminants in the environment should be followed, using as an indicator the ***tissue content of contaminants*** in, for example, *Mya*. This is a pressure indicator of a specific contaminant, which is not complicated to measure, integrative, and scientifically valid.

Physical ecosystem component indicators
In the physical component of the ecosystem, the sediment structure could change due to any of the four issues. Currents are changing following habitat modifications and climate change, whereas the turbidity is especially relevant for pollution and habitat modifications.

A change in the sediment or substrate could be detected by ***granulometry***. The granulometry to be used is measured using a block of sediment (mixed layers). Although the precision and detail of information would be much higher if separate layers were examined, the cost-effectiveness of the former indicator is obviously higher. The amount of organic matter is another relevant indicator (***% organic matter***), which is simple to obtain. The ***turbidity*** (nephelometric turbidity units) is also easy to obtain and timely.

Additional factors to be followed for the fisheries issue are temperature, the occurrence of storms, ice scouring, and the amount of primary production (large-scale measures of primary production, e.g. by remote sensing of chlorophyll *a*). For the pollution issue, it is the temperature of air and water, precipitation, and the occurrence of storms that must be followed. For habitat modifications, precipitation and storms as well as salinity need to be followed. Finally, under the climate change issue, additional factors to measure are air temperature, occurrence and strength of storms, ice cover, and erosion.

B. Pressure indicators

As a pressure indicator for fisheries, we suggest the use of the ***number of people collecting clams per area per day***. For pollution we suggest the ***number of inhabitants in the adjacent zones***, ***number of industries***, and ***extent of agriculture***, combined with information on

wastewater cleansing. As pressure indicators for habitat modifications, we suggest the ***number of hydroelectric dams***, ***their size***, and the ***change in outflow from rivers*** caused by the dams. For climate change, ***emission of*** $\boldsymbol{CO_2}$ estimates the pressure exerted by human activities. However, this includes a long time lag and an additional indicator such as the ***mean annual temperature*** may be used as a pressure indicator.

5.4.2. Comments on the indicator set

There is a large overlap in the indicators chosen for the four major issues. It can be noted that as we move up in the biological part of the organisational hierarchy, the indicators selected are more often identical due to the integrative nature of higher organisational stages.

The majority of the selected indicators can be placed in the category of compliance indicators (as soon as they are associated with appropriate objectives). Only enzyme activity, gonadosomatic index, and burial depth are early-warning indicators, whereas granulometry and turbidity may function as a diagnostic indicator. The additional factors to be followed can also be used as diagnostic indicators. Thus, the indicator set conforms to the prerequisites mentioned in the indicator section.

6. CONCLUDING REMARKS

In this report we have developed a framework for selecting indicators to evaluate and monitor the health of marine biological communities. We have first dealt with the concept of ecosystem health and from there developed an ecosystem approach to help us obtain indicators. This ecosystem approach includes the identification of major issues affecting a community and the properties of each ecosystem level that are affected by the major issues. Then we suggested possible indicators for the properties and after an evaluation of their fulfilment of indicator criteria, we selected a final set of indicators.

One of the advantages of the approach is that it stresses the need for a prioritisation at different stages: issues, organisational level, properties, and indicators. It allows entering into detailed considerations while the overview is maintained, and it aims at limiting the number of indicators needed to cover a topic. Finally, the approach encourages a question-driven monitoring, since one is forced to put forward hypotheses when setting objectives and benchmarks.

This report did not develop presentation schemes for the values observed in the indicator set. Among the most widely used methods is the “traffic light approach” (Caddy 1998), where different colours signal how the indicator values correspond to the objectives. Another prominent method of representation using a composite index to summarise the index values for each indicator into one value is the index of ecological integrity (Karr et al. 1986). A third one, the amoeba method, uses a circle sliced up according to the number of indicators in the set, and the objectives are represented by the radius of the circle. Any indicator value that exceeds the objective also exceeds the radius of the circle (Brink 1991; De Zwart *et al.* 1995).

Throughout the report, we have used the *Mya–Macoma* community as an example of the application of the approach. However, the major issues ecosystem approach is also applicable to other biological communities and to other scales of the ecosystem. It is conceivable, for example, to develop a monitoring framework for a larger marine area using the same approach. In that case, the issues would more likely have an impact at the highest level of the ecosystem.

The set of indicators we have presented for the *Mya–Macoma* community has not yet been tested. Although the validity of many indicators is well known, the performance of the entire set should be tested, using the multiple-sites approach with different degrees of impact. Studies are currently being conducted in the Quebec Region on the community to examine the validity of a number of indices.

For the *Mya–Macoma* community, the objectives and benchmarks for each indicator should eventually be determined. Certain objectives may vary according to site-specific variations, even if the overall objectives remain the same within a larger area. However, the association of benchmarks to each indicator can be done on a more general basis. In fact, the establishment of benchmarks in any ecosystem or community context is likely to be the most difficult part of the overall monitoring initiative. It depends upon detailed scientific knowledge of the attributes or processes implied.

The present selection of indicators is limited by our knowledge of the ecosystem and the benthic communities in the St. Lawrence Estuary and Gulf. As a monitoring program would progress and more knowledge accumulates, the number of indicators could perhaps be diminished. It is also possible that the indicator set would benefit from other adjustments. However, these changes should be kept to a minimum in order to ensure the continuity of the monitoring.

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APPENDIX

Some important terms used in conservation and environmental assessment

Biodiversity: "The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (Convention on Biodiversity [Anon. 1992]).

Ecosystem health: within the MEQ framework: "an ecological system is healthy (and free of "stress syndrome" if it is stable and sustainable) if it is active and maintains its organization and autonomy over time and is resilient to stress" (Costanza 1992).

Ecological or biological integrity: The capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organisation comparable to that of natural habitat of the region (Karr and Dudley 1991; Angermeier and Karr 1994).

Ecological quality: a description of ecological quality of marine ecosystems is: "an overall expression of the structure and function of the marine ecosystem taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions, including those resulting from human activities" (TemaNord 1999). To obtain an evaluation, this description is compared to "a desired level of ecological quality relative to a reference level" (TemaNord 1999).

Ecological sustainability: "The use of components of biological diversity in a way and at a rate that does not lead to long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations" (Convention on Biodiversity [Anon. 1992]).

Littoral zone: the littoral zone includes the intertidal zone and the infralittoral zone, extending from the highest high water spring tide down to the lower limits for macrophytic growth, an approximate 1% light level (Mathieson and Nienhuis 1991).

Naturalness: "Natural can be defined as the way the system in question would function (or would have functioned) in the absence of humans"; "nature as meaning free of human impact" (Anderson 1991).

Pristine area: This refers to an area untouched by civilisation, an area in its virgin or original state. Pristine conditions can be found in all developmental or successional states and thus does not refer to a particular stage of maturity.

Quality: This term is often used in relation with water quality, where it is usually understood as water purity, whereas the biological aspects have been excluded. Suter (1993) suggests using quality instead of ecosystem health to stress to the subjective nature of assessments, but does not specify "quality" further. The term "nature quality" has been used in some cases (Nygaard *et al*. 1999) but has not gained wider application.

Wilderness: "….huge, roadless, essentially unmanaged areas;… wilderness provides a standard of healthy, intact, relatively unmodified land" (Noss 1991).