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Indicators and Thresholds for Use in Assessing Shellfish Aquaculture Impacts on Fish Habitat

Indicateurs et seuils pour l'évaluation des effets de la conchyliculture sur l'habitat du poisson

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ABSTRACT

The purpose of this research document is to provide science advice needed to allow DFO Habitat Management to make and justify management decisions related to the potential harmful alteration, disruption and destruction (HADD) of fish habitat by shellfish aquaculture. The overall goal of this exercise and of our recommendations is to promote the avoidance and mitigation of a HADD. Our specific objectives were the following:

1. identify, evaluate and make recommendations regarding a range of quantitative indicators (measures of habitat and ecosystem status) that could be used to monitor for potential shellfish aquaculture effects; and
2. provide science-based decision support for the development of an environmental monitoring framework, based on identification of predetermined impact limits (operational thresholds) intended to trigger management actions.

A wide range of ecosystem and habitat status indicators and methodological approaches were identified to support industry management and each was initially screened based on habitat impact predictions and observations. Selected indicators were classified based on associated strengths and weaknesses using predefined criteria, including: availability of operational thresholds; regulatory needs; cultured species; scales of impact addressed; cost/benefit; and the needs of responsive management. A habitat assessment framework is recommended for shellfish aquaculture that addresses the need for a consistent and transparent decision-making approach that is science-based, and reflects both fish habitat and ecosystem concerns.

The highly diverse Canadian shellfish aquaculture industry (e.g. species cultured, husbandry method, and stocking density) and regional differences in environmental impact risks (related primarily to geographic and hydrodynamic factors) were identified as important considerations for our evaluation of shellfish aquaculture impact assessment options. Recommendations are made towards establishment of an environmental monitoring framework that incorporates sufficient flexibility to be of use in a wide range of environmental settings and that is both effective and practical for current aquaculture operations that range from less than 0.5 to 500 hectares.

A primary recommendation of this report is that habitat assessments could be based on a tiered approach that recognizes that an increased risk to fish habitat requires an increase in monitoring effort. Various levels of monitoring could be triggered based on assessments of environmental sensitivity and risk (e.g. dispersive vs. depositional environment and presence of sensitive habitat), the nature of the operation (e.g. size, species and husbandry), and previous measurement and verification of environmental impacts. Inherent within the recommended framework is that ongoing monitoring programs could be continually adaptive to changes in our state-of-knowledge on potential environmental impacts, indicators and related methodologies. It is important to maintain an ability to add or remove indicators to monitoring programs based on sound science.

The recommended multi-tiered impact assessment approach addresses the potential for benthic marine habitat impacts in the immediate vicinity of each shellfish aquaculture lease

and it therefore parallels science recommendations for finfish aquaculture monitoring in Canada. Scientifically defensible thresholds are available for benthic biogeochemical indicators (sulfides and redox potential) and these could be used to define the hypotheses that need to be addressed in an operational monitoring program. Effective measures are also available for mitigating benthic organic enrichment impacts, and these can be linked to the operational thresholds incorporated in a responsive management framework.

Ecosystem-level interactions with dense shellfish aquaculture populations are more complex than for finfish culture and many potential and observed effects on fish habitat cannot be assessed using only site-specific benthic habitat indicators. Measurements with selected far-field impact indicators are needed under certain conditions to compliment benthic operational monitoring. The inability to fully define quantitative operational thresholds for many valid and highly relevant indicators of habitat and ecosystem status (particularly those describing the structure and dynamics of pelagic habitat), owing to present gaps in our knowledge of ecosystem dynamics, should not preclude their potential use. Surveillance sampling programs based on water column parameters are needed under conditions where environmental impact assessments and ongoing monitoring data indicate a relatively high risk that bay-scale impacts will occur. Of particular importance is the need to assess the impacts of longline mussel culture operations on suspended particle concentrations and distribution and the pelagic food web (micro-flora and fauna) in extensively leased coastal embayments. Industry shellfish stocking information for all farms within a management area is considered fundamental to assessments of shellfish aquaculture impacts on fish habitat. The use of sound science practices is required for the design of monitoring programs (statistically valid sampling approaches) and for the analysis of habitat status indicators and data (e.g. quality assurance/quality control).

RÉSUMÉ

Le présent document de recherche a pour but de fournir un avis scientifique afin de permettre à Gestion de l'habitat du MPO de prendre des décisions de gestion (et de les justifier) concernant la destruction, la détérioration ou la perturbation (DDP) de l'habitat du poisson par la conchyliculture. Le but général de cet exercice et de nos recommandations est d'éviter ou d'atténuer la DDP de l'habitat. Nos objectifs particuliers sont les suivants :

1. relever et évaluer un éventail d'indicateurs quantitatifs (mesures de l'état de l'habitat et de l'état de l'écosystème) qui pourraient être utilisés pour surveiller les effets potentiels de la conchyliculture et faire des recommandations à cet égard;
2. fournir une aide scientifique à la décision pour l'élaboration d'un cadre de suivi environnemental fondé sur l'établissement de limites d'effets prédéterminés (seuils opérationnels) destinées à initier des mesures de gestion.

Un grand nombre d'indicateurs de l'état de l'écosystème et de l'habitat et d'approches méthodologiques ont été recensés pour soutenir la gestion de l'industrie. Chaque indicateur a été sélectionné a priori d'après les prévisions et les observations liées à l'effet sur l'habitat. Les indicateurs retenus ont été classés en fonction de leurs forces et de leurs faiblesses à l'aide de critères prédéfinis, notamment l'existence de seuils opérationnels; les exigences réglementaires; les espèces élevées; les degrés d'effets pris en considération; le ratio coût/avantages; les exigences en matière de gestion adaptée aux besoins. Dans le cas de la conchyliculture, nous avons recommandé un cadre d'évaluation de l'habitat qui tient compte de la nécessité d'opter pour une approche décisionnelle cohérente, transparente et à fondement scientifique qui reflète à la fois les préoccupations liées à l'habitat du poisson et à l'écosystème.

Parmi les points importants pris en considération dans le cadre de notre évaluation des options relatives à l'étude des impacts de l'industrie conchylicole, nous avons relevé la grande diversification de cette industrie au Canada (p. ex. espèces élevées, méthodes d'élevage, densité de peuplement) de même que des différences régionales dans les risques d'effets environnementaux (liés principalement à des facteurs géographiques et hydrodynamiques). Nous recommandons d'établir un cadre de surveillance de l'environnement suffisamment souple pour être utilisé dans une variété de conditions environnementales et qui est efficace et pratique pour les exploitations aquicoles actuelles, dont la taille varie de moins de 0,5 à 500 hectares.

L'une des principales recommandations de ce rapport est que les évaluations de l'habitat soient axées sur une approche multi-niveaux qui reconnaît qu'un risque accru pour l'habitat du poisson requiert, par conséquent, un effort de surveillance accru. Divers degrés de surveillance peuvent être appliqués en fonction des évaluations de la vulnérabilité de l'environnement et du risque pour l'environnement (p. ex. milieu dispersif ou milieu de dépôt et présence d'un habitat vulnérable), de la nature de l'exploitation (notamment la taille, les espèces et les élevages) et des effets environnementaux observés par suite de mesures et de vérifications. Le cadre recommandé prévoit que les programmes de surveillance continue puissent être adaptés régulièrement aux changements dans nos connaissances des effets environnementaux potentiels, des indicateurs et des

méthodologies connexes. Il est important de pouvoir ajouter ou enlever au besoin des indicateurs aux programmes de surveillance si l'on dispose de justifications scientifiques solides à cet égard.

L'approche d'évaluation progressive des effets recommandée tient compte des possibilités d'effets sur l'habitat marin benthique situé à proximité immédiate de chaque concession conchylicole et, par conséquent, correspond aux recommandations scientifiques relatives à la surveillance de la pisciculture au Canada. Des seuils défendables sur le plan scientifique sont disponibles pour les indicateurs biogéochimiques du milieu benthique (sulfures et potentiel d'oxydoréduction), et ces seuils pourraient être employés pour définir les hypothèses qui doivent être étudiées dans un programme de surveillance opérationnelle. Des mesures efficaces sont également disponibles pour atténuer les effets de l'enrichissement organique du milieu benthique, lesquelles mesures peuvent être associées aux seuils opérationnels intégrés dans un cadre de gestion adaptée aux besoins.

Les interactions écosystémiques avec les populations conchylicoles denses sont plus complexes que celles avec les populations piscicoles. Or, de nombreux effets sur l'habitat du poisson potentiels et observés ne peuvent être évalués uniquement à l'aide les indicateurs de l'habitat benthique propres au site. Pour compléter la surveillance opérationnelle du milieu benthique, nous devons, sous certaines conditions, recourir à des mesures intégrant des indicateurs des effets à distance sélectionnés. Les lacunes actuelles dans notre connaissance de la dynamique de l'écosystème font en sorte que nous sommes incapables de définir entièrement les seuils opérationnels quantitatifs pour bon nombre d'indicateurs valides et très pertinents de l'état de l'habitat et de l'écosystème (surtout ceux qui décrivent la structure et la dynamique de l'habitat pélagique), ce qui ne devrait pas pour autant exclure l'utilisation potentielle de ces indicateurs. Nous devons recourir à des programmes d'échantillonnage de surveillance reposant sur les paramètres des colonnes d'eau lorsque les évaluations des effets environnementaux et les données de surveillance continue indiquent un risque relativement élevé que des effets se produisent à l'échelle d'une baie. Il est particulièrement important d'évaluer les effets des exploitations mytilicoles en boudins sur les concentrations et la répartition des particules en suspension de même que sur le réseau trophique pélagique (microflore et microfaune) que l'on rencontre dans des enfoncements côtiers où les concessions abondent. Les données de l'industrie sur le peuplement conchylicole de toutes les exploitations situées dans une zone de gestion servent de fondement aux évaluations des effets de la conchyliculture sur l'habitat du poisson. Il faut employer des pratiques scientifiques éprouvées pour concevoir les programmes de surveillance (approches d'échantillonnage statistiquement valides) et pour analyser les données et les indicateurs relatifs à l'état de l'habitat (p.ex. assurance et contrôle de la qualité).

1.0 INTRODUCTION

DFO Habitat Management is seeking advice from DFO Science on potential monitoring approaches and requirements for shellfish aquaculture operations. This working paper is one of five science advisory documents that address the issue of marine shellfish aquaculture-environment interactions (Appendix 1). The following Terms of Reference for science advice were provided for consideration by the authors under the theme of "Indicators and Thresholds for Use in Assessing Shellfish Aquaculture Impacts on Fish Habitat" (Paper #2):

- benthic, pelagic and shellfish performance indicators and thresholds, including near, far-field and cumulative effects; and
- monitoring frameworks for assessing fish habitat effects of shellfish aquaculture and methodologies, including case studies.

All authors were instructed to consider how regional and operational differences may impact the applicability of tools and approaches used to assess shellfish aquaculture effects on fish habitat.

Although the culture of shellfish is an important and well established industry in Canada, research programs to investigate the impacts of aquaculture have, until recently, focussed largely on finfish sea-cage culture. The preparation of this document follows in the wake of the finfish aquaculture National Advisory Process that had many similar objectives. It is important to emphasize that there are very significant differences between the culture of fish and shellfish. Most farmed fish are carnivorous and depend on the addition of food to the environment, whereas shellfish are herbivorous and eat microalgae and detritus that are naturally available in the water. Unlike finfish aquaculture, bivalve culture requires no addition of organic matter; their food is supplied by the environment and their wastes return nutrients and minerals to the ecosystem. Shellfish culture is therefore intricately and inextricably linked to its environment.

Unlike finfish culture, industry husbandry practices for rearing various shellfish species include a wide array of options including floating cages, bottom cages, bottom plots, suspended collectors, rafts, tables, longlines, and poles. Potential marine shellfish culture sites are not nearly as limited by hydrodynamic conditions as is the case for finfish; suitable shellfish aquaculture sites span a wide spectrum of habitats from the intertidal zone to shallow and deep coastal embayments. This greater range of husbandry techniques and habitats translates into a greater complexity of potential environmental interactions. Determining the net impact of shellfish aquaculture on fish habitat, community structure and ecosystem productivity is complex and requires an objective and holistic approach. Important ecological interactions between land-use and cultured shellfish have also been identified, further adding to the complexity of this issue (see Working Paper #4).

The extensive development of shellfish culture operations and the current requirement to conduct environmental assessments under the *Canadian Environmental Assessment Act* have resulted in a closer scrutiny of the impacts of this industry on the environment.

Concerns have been raised about the possible effects of extensive shellfish culture operations on coastal marine ecosystems and the related risks to the ecological functioning and sustainability of these regions. As the area under culture continues to expand and stocking levels increase, the need to evaluate these risks becomes imperative. In response to these concerns, the DFO State-of-Knowledge Initiative (Program for Sustainable Aquaculture) was established to provide peer-reviewed reports on the status of scientific knowledge on the potential environmental effects of shellfish aquaculture, with emphasis on knowledge relevant to Canada (Cranford et al., 2003). A subsequent review of potential shellfish aquaculture impacts was carried out by the ICES Working Group on Marine Shellfish Culture (ICES, 2004). The results of these initiatives provided a basis for our identification of monitoring needs, and for our evaluation of potential monitoring indicators and approaches. Additional knowledge and expertise stems from the various authors' participation in several recently completed research programs on the local and ecosystem-scale impacts of bivalve culture in Canada (see Section 1.1.5).

It is generally acknowledged that the culture of bivalve molluscs may have a wide range of impacts on the habitat and community structure of coastal marine ecosystems. Such factors as the type and extent of culture activities, and the local environmental conditions, are critical to determining the degree of impact and the net effect on fish habitat. For example, certain culture methodologies and practices have been identified as having a greater potential for environmental impact than others (ICES 2004). This disparity among the effects of different industry practices is primarily related to variations in stocking density per unit area and differences in environmental sensitivity or ability to absorb the impacts of bivalve culture.

Of particular concern is the longline culture of mussels which is believed to have a relatively high potential for local and bay-wide impacts (ICES, 2004). This rearing technique involves the deployment of densely packed mussel cohorts throughout much of the water column, resulting in relatively high stocking densities per unit area and volume compared with other species currently under culture in Canada. At present mussel culture constitutes approximately 80% of the shellfish aquaculture landed value in Canada, with a large fraction of this industry based in Prince Edward Island (PEI). Mussel leases in extensively cultured embayments in PEI may occupy up to 50% of the low-tide surface area. However, even under these dense culture conditions, impacts may not be significant if local hydrographic conditions permit the rapid flushing of lease areas and the widespread dispersion of feces.

The following introductory subsections briefly summarize: (a) the major findings of the above previous reviews on shellfish aquaculture/ecosystem interactions (supporting references are provided in the published reviews, except where indicated); (b) principles and "state-of-the-art" approaches to environmental monitoring; and (c) the criteria employed by the authors for evaluating potential fish habitat indicators. This background information is followed by our analysis of potential indicators and thresholds for managing the impacts of shellfish aquaculture on water column (pelagic) and seabed (benthic) habitat. It was not our intention to provide an in-depth review of international research on the potential impacts of shellfish aquaculture, but published and unpublished research papers are presented to substantiate our assessments and to support our recommendations on appropriate indicators and thresholds.

1.1. Summary of shellfish aquaculture impacts

Cultured shellfish (narrowly defined in this paper to include only bivalve molluscs) and their associated rearing structures have the potential to impact the environment in positive and negative ways. Four basic areas of concern are the effects of bivalve culture on: (1) suspended particles, particularly in terms of food resources; (2) sediment geochemistry/benthic habitat; (3) nutrient cycling; and (4) benthic and pelagic population dynamics/community structure. The potential effects summarized below are related to the four basic processes illustrated in Figure 1.1 (shellfish filter-feeding, feces production, excretion and harvesting).

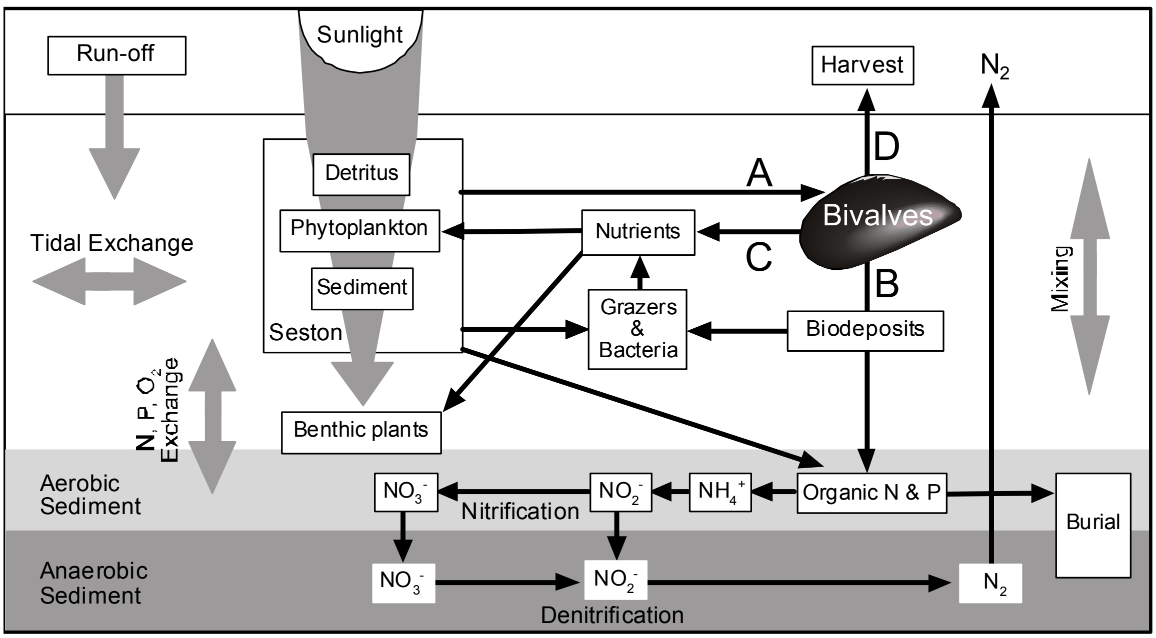


Figure 1.1. Conceptual diagram of shellfish (bivalve) aquaculture interactions in coastal ecosystems related to: (A) the removal of suspended particulate matter (seston) during filter feeding; (B) the biodeposition of undigested organic matter in feces and pseudofeces; (C) the excretion of ammonia nitrogen; and (D) the removal of materials (nutrients) in the bivalve harvest.

1.1.1 Potential impacts on particle dynamics / food resources

Bivalve filter feeders depend on a supply of particulate food resources, primarily phytoplankton and detritus, but including some auto- and heterotrophic picoplankton and zooplankton. Food particle depletion has been documented within bivalve culture sites, including sites in Nova Scotia (NS) and PEI. Bay-scale particle depletion, both predicted and observed in extensively leased PEI embayments, likely impacts food webs within the system. A summary of potential impacts includes:

- Increased competition for food resources: bivalve filter feeders have a high filtration capacity, such thathigh density cultivation may deplete the resident phytoplankton and seston with negative effects on other resident filter-feeders such as zooplankton and natural populations of bivalve shellfish. Food resource depletion may result in the community becoming dependent on the tidal input of offshore phytoplankton.
- Increased biodeposition: egestion of pseudofeces and feces particles (aggregation and sedimentation) increases the rate of organic particle deposition to the bottom. This may represent an additional food source for benthic organisms or it may lead to habitat degradation (see Section 1.1.2).
- Shifts in the particle size spectra/phytoplankton composition associated with bivalve selective feeding behaviour could negatively impact other filter-feeding populations.
- Alteration of local particle aggregation and sedimentation rates by reducing particle concentration or by disruption of natural particle aggregation processes.
- Depletion of excess phytoplankton under eutrophied conditions: bivalve particle feeding may effectively reduce the impacts of coastal nutrient enrichment from land-use. For example, bivalve grazing may increased water clarity, thereby expanding the depth range and biomass of macroalgae.
- Increased sedimentation due to the alteration of current flow patterns as a result of physical interference posed by shellfish installations may impact benthic communities.

1.1.2. Potential impacts on the benthic habitat

Shellfish feces and pseudofeces contain organic matter (15 to 50% organic content) which may cause benthic enrichment effects. By diverting fine suspended organic particulate matter to the seabed through ingestion and egestion, suspended culture may impact sediment geochemistry and benthic habitat productivity. The degree of organic enrichment is closely related to the hydrodynamics of the system, and hence the potential for impact is highly site specific. Estuaries identified as having the greatest risk of biodeposition effects are generally shallow, have a relatively small tidal exchange and have a high percentage of the total estuarine volume under culture. Many intensively leased mussel aquaculture embayments in PEI can be included in this high-risk category (Cranford et al., 2003). A summary of potential impacts includes:

- Recycling of organic biodeposits increases the oxygen demand in the sediments, potentially generating an anaerobic environment that promotes sulfate reduction. This is comparable to the situation under finfish cages.
- Increased sulfide levels and associated habitat degradation lead to a reduction in benthic species abundance and diversity and shifts in benthic community composition.

- Enhanced abundance of fauna associated with low level organic enrichment (i.e. increased food levels for infaunal deposit-feeders and predators).
- Oxygen depletion in the water column. Observations of hypoxic/anoxic conditions resulting from the high Biological Oxygen Demand of biodeposits are limited, but indicate that effects may be limited in time and space with greatest effects localized near the seabed.
- Shellfish and epibiont (fouling organisms) fall-off from suspended culture may contribute to the negative impacts of organic loading and/or provide an additional food source for benthic predators.
- Reduction in macrophyte communities due to increased shading of the bottom directly below bivalve culture.

1.1.3. Potential impacts on nutrient dynamics

Shellfish excrete ammonia and other nutrients and, if excreted in sufficient quantities under some conditions, can significantly impact coastal nutrient dynamics. Specific impacts include:

- Increased ammonia levels may promote phytoplankton production and/or alter phytoplankton species composition which may in turn affect grazer species composition and abundance.
- Increased rates of nitrogen cycling in coastal regions due to the more rapid deposition of suspended organic matter and the subsequent nutrient regeneration in sediments.
- Increase in local nutrient availability as less material is exported from the system.
- More frequent algal blooms due to the greater availability of nutrients. There has been much speculation on the contribution of bivalve culture to the incidence of harmful algal blooms (HABs), but there is no evidence supporting a direct link.
- Harvesting of bivalves contributes to the removal of excess nutrients from eutrophicated coastal systems, but effectively represents a net loss from nutrient-limited systems.

1.1.4. Potential impacts on population dynamics and community structure

The suspended and bottom culture of bivalves increases the surface area available for attachment and grazing by other species, and provides refugia from physical stress (currents and waves) and predation. Impacts on various populations include:

- Increased recruitment of biofouling filter-feeding epibiont species such as sponges and ascidians. These filter-feeding epibionts may have effects that are comparable to those of bivalves, such as removal of microalgae, biodeposition, and excretion.
- Decline in zooplankton populations through food resource competition and direct ingestion by the bivalves and/or their associated biofouling populations may alter pelagic food webs.
- Mussel and epibiont falloff from suspended culture may represent an additional food source for benthic predators such as crabs, lobsters and demersal fish species and may also alter benthic community structure.

- Predation of cultured bivalves by predators attracted to the culture area may cause increased predation on wild species.
- Mussel seed and/or the fouling associated with bivalve culture may provide an important food source for bird populations such as eider ducks.
- Spawning of cultured bivalves may enhance recruitment of local bivalve populations.
- Human activity associated with bivalve culture may disrupt local bird populations and other marine fauna.
- Hydrocarbons and wastes introduced to the environment from increased boating activity in culture areas can have deleterious ecological effects, but are addressed in industry Codes of Practice.

1.1.5 Supplementary contributions from recent research in Canada

Worldwide research has identified a wide array of potentially positive and negative effects of shellfish aquaculture on fish habitat and ecosystem structure and function (outlined above). In an attempt to further this research effort at aquaculture sites in Canada, the DFO Program for Sustainable Aquaculture (PSA) provided directed funding for environmental science research at DFO, and established the Aquaculture Collaborative Research & Development Program (ACRDP) to foster growth of a sustainable and internationally competitive aquaculture industry and to increase public confidence in aquaculture. Several environmental and biological science research projects conducted under the PSA were recently completed, or are approaching completion. These projects were designed to provide information relevant to the development and implementation of effective area-wide strategies that will promote a sustainable shellfish aquaculture industry. The authors' direct participation in the research projects outlined in Table 1.1 was critical for the development of the science-based advice provided in this paper.

1.2. Fundamentals of environmental effects monitoring

The purpose of any industry monitoring program is not exclusively to document the temporal and/or spatial scale of environmental impacts, but is to *prevent* potentially significant negative impacts from occurring through the responsive management of industry practices. Monitoring programs are designed to encourage the avoidance and mitigation of any harmful alteration, disruption or destruction of fish habitat. Environmental effects monitoring (EEM) is a central component of environmental protection and management strategies designed to minimise the consequences of anthropogenic activities (GESAMP, 1991). Frameworks for environmental management generally consist of a linked series of activities that identify, critically evaluate and address predictions of potential environmental effects (Cranford and Lee, 2005). Environmental impact predictions are inherently subject to some degree of uncertainty as

Table 1.1. Recently completed and ongoing research projects conducted by the authors that contributed to the formulation of scientific advice on potential indicators and thresholds for assessing shellfish aquaculture effects.

Title and Description	**Principle Investigators**	**Funds Source Duration**
Integrated ecosystem studies for modelling mussel aquaculture /ecosystem interactions. Develop methodologies and technologies and investigate the ecosystem-level effects of mussel aquaculture. Integrate scientific data on the consequences of mussel culture to ecosystem structure and function through the use and predictive power of ecosystem modelling. Use holistic ecosystem approaches to understand and predicting the overall effects of multiple interactions of bivalves within dynamic coastal ecosystems. Understand the relative impact of different mussel culture methods and hydrographic regimes on ecosystem structure and function.	P. Cranford, B. Hargrave, P. Strain, S. Bates, J. Grant, C. Bacher, E. Horne, P. Kepkay, T. Milligan, G. Harrison, B. Li, G. Bugden, M. Dowd M. Fréchette, P. Archambault, S. Robinson, M. Ouellette	$860,000 ESSRF 2001- 2004
Development and evaluation of standardized monitoring and data acquisition systems for the management of mollusc culture in Atlantic Canada. Develop a series of standardized protocols for monitoring mussel growth, survival and yield in relation to husbandry and environmental conditions on several mussel grow-out sites. The performance of mussels (growth and survival) is influenced by husbandry and environmental conditions.	L.A. Comeau	$390,400 ACRDP 4 years
Environmental carrying capacity of mussel culture: Evaluation of biodeposition of macro and micro particles and their effects on the environment. Develop a predictive model of the temporal and spatial pattern of sedimentation on mussel culture sites and to assess the effect on the structure of the benthic assemblages. Establish the relationship between the sedimentation of micro-particles associated with mussel culture and the structure of the benthic communities. This model will be aligned with a more general model on the carrying capacity.	P. Archambault, C. McKindsey	$301,800 ACRDP 2003- 2006
Monitoring fishes and macroinvertebrates to determine indirect influence of bivalve aquaculture on ecosystem productivity. This research will test the general hypothesis that bivalve culture sites increase the productivity of fish and macroinvertebrates. An increase in the productivity of this component of the ecosystem may offset some of that commonly believed to be lost due to the presence of aquaculture. The proposed research will help us better understand the mechanisms that increase the productivity of fishes and macroinvertebrates, thereby justifying the evaluation of their abundances as a key tool for environmental assessments and monitoring of aquaculture sites.	C. McKindsey, P. Archambault, T. Landry	$119,363 AquaNet 2004- 2006
Impact of suspended and off-bottom eastern oyster culture on the benthic environment in eastern Canada.	A. Mallet, C. Carver	$55,000 ACRDP 2002- 2003

Table 1.1. Continued.

Environmental requirements for sustainable shellfish aquaculture The objective of this project was to assess the relationships among key environmental variables and the suitability of sites to support sustainable shellfish aquaculture. Research was focused on the impact of high-density shellfish culture on the pelagic and benthic ecosystem processes and used a combination of empirical and modeling approaches to evaluate the direct and indirect effects among variables on shellfish aquaculture.	M. R. Anderson D. Deibel R. B. Rivkin R. J. Thompson	\$512,300 AquaNet 2000-2004

a result of gaps in knowledge, the natural variability of ecosystems and the influences of unforeseen factors. This uncertainty is addressed by operational monitoring programs that ensure that actual effects do not exceed the predictions and are within predetermined acceptable limits (operational thresholds). It is generally accepted that management responses to monitoring data should be timely and follow a predetermined course of action. Furthermore, the management framework includes feedback-loops among prediction, monitoring and management responses that serve to strengthen our ability to protect and manage fish habitat and marine ecosystems.

Environmental monitoring has been defined as "the regular collection, generally under regulatory mandate, of biological, chemical, or physical data from predetermined locations such that the present status and any ecological changes attributable to aquaculture can be quantified" (GESAMP, 1996). EEM programs are designed around the testing of hypotheses that are based on concrete program objectives, including:

a) establishing pre-development (background) environmental conditions to aid in the design of subsequent EEM;
b) determining natural trends in the environment and degree of variation during industry operations;
c) ensuring conformity with regulations and specific management requirements (i.e. compliance monitoring);
d) ensuring that waste concentrations and distributions and environmental effects do not exceed predictions made in environmental impact assessments (includes testing of impact assessment tools, such as predictive models and other risk assessment methodologies);
e) identifying spatial and temporal trends in discharge waste distributions and biological effects, and ensuring that contaminant concentrations and effects are not significantly greater than acceptable limits (operational thresholds);
f) relating physical (e.g. current speed and direction), geological (e.g. sediment grain size) and chemical parameters (e.g. nutrients, hydrogen sulfide) to habitat effects;
g) assessing the effectiveness of the environmental management framework, including the effectiveness of any mitigation measures imposed;
h) monitoring rates of natural recovery and/or the effectiveness of mitigation programs at impacted sites;
i) providing feedback for assessing the adequacy of regulatory standards and/or guidelines imposed on industry for environmental conservation and protection;
j) understanding and delineating cause-effect relationships; and

k) helping to identify future research priorities.

EEM objectives are assessed on the basis of observations of the spatial extent and magnitude of site-specific environmental effects, while taking into account the presence of other potential sources of effects. An integrated monitoring approach is routinely undertaken during industry monitoring programs where a suite of environmental impact indicators is monitored in the receiving environment, including reference sites, and the environmental performance of the industry is based on the accumulated evidence. This "weight-of-evidence" approach is desirable over situations where decisions that could result in operational or regulatory actions are based on more limited investigations (Cranford and Lee, 2005).

Surveillance is another scientifically defensible approach towards the ongoing assessment of anthropogenic impacts. Unlike operational monitoring, which requires the testing of specific predictions or hypotheses (e.g. predetermined impact thresholds), surveillance does not address specific operational thresholds, but instead attempts to determine if there are detectable differences between aquaculture and control sites, or significant changes over time that cannot be attributed to natural variations. This approach allows environmental concerns to be addressed where the science has yet to advance to a point where definitive habitat threshold conditions may be established. It is therefore founded on a precautionary strategy (Gray et al., 1991).

Monitoring programs should be conducted in a manner that adheres to a wide array of basic scientific principles, including:

- Use sound science as a foundation to methodology.
- Incorporate useful, applicable, and easily understandable performance indicators.
- Provide trigger mechanisms (thresholds) to alert regulators and stakeholders of the need for changes in aquaculture practices.
- Link thresholds to defensible and effective mitigation measures.
- Implement standard methodologies and data quality assurance and control standards.
- Standardize documentation of environmental conditions at aquaculture sites.
- Provide a proper system of data collection, quality assurance, archiving, and retrieval that is transparent and accessible.
- Provide flexibility to include additional or alternative indicators when proven to be more effective (e.g. cost and performance) than current methodologies, or are shown to effectively address emerging environmental concerns.

Although no standard regional approaches to EEM currently exist for shellfish aquaculture in Canada, a few programs are active and these have been considered in the preparation of this report. Of particular note is the environmental management plan for the Bounty Bay Shellfish Inc. and 5M Aqua Farms mussel aquaculture operation in St. Ann's Harbour, Cape Breton, NS (Stuart, 2003). This is the largest shellfish culture operation in Canada (490 ha) and monitoring has been ongoing annually since 2000, when collection of baseline information first began. Also of significance to our efforts is the content of a report prepared by the Aquaculture Association of Nova Scotia (2002) entitled "*Design of the Environmental Monitoring Program for the Marine Aquaculture Industry in Nova Scotia*". These reports have had a significant influence on several of our recommendations.

1.3. Considerations for indicator identification and selection

A consistent and objective approach was used to identify and assess potential indicators and thresholds. This exercise was somewhat constrained in scope by the Terms of Reference provided (Appendix 1). As stated above, our focus on identifying indicators and thresholds was solicited primarily to address the issue of "harmful" alterations to fish habitat. Although a fully integrated ecosystem-based approach was not taken, ecosystem effects were considered at all levels of planning, owing to the complexity of shellfish /ecosystem interactions. Our recommendations are therefore highly relevant to forthcoming ecosystem management initiatives.

It is not solely the responsibility of scientists to determine what constitutes a harmful alteration, disruption or destruction (HADD) (or, in the terminology of ecosystem-based management, an "ecosystem operational objective"), as there are important socio-economic dimensions to any such discussion. Our role as scientists is to provide advice from the perspectives of the ability to: (1) characterize environmental conditions and aquaculture impacts (indicator identification); and (2) identify factors that could indicate the potential consequences to fish habitat and productivity resulting from changes in this status (threshold recommendations). This requires that we not consider the potential consequences to industry or society stemming from our recommendations.

The selection and evaluation of potential environmental indicators is based on the principles outlined in the following subsections. After an initial screening of the attributes and potential applications of a broad spectrum of potential indicators, a number of indicators were identified for presentation and discussion in this document. Each author prepared draft documents on specific indicators and thresholds related to their research experiences and specific fields of expertise. These were discussed at a working group meeting on 13 January, 2006. After considering comments by the group, their revised texts were incorporated into this working paper. Group and individual discussions among authors also helped to build a consensus on the recommended monitoring framework.

1.3.1 Regulatory/management requirements

For the purposes of this document, and as instructed in the NAP Terms of Reference (Appendix 1), indicators of various industry effects need to be linked to the harmful alteration, disruption or destruction (HADD) of fish habitat (Section 35 of the *Fisheries Act*). The broad definitions of "fish" (parts of fish, shellfish, crustaceans, marine animals and any parts of shellfish, crustaceans or marine animals, and the eggs, sperm, spawn, larvae, spat and juvenile stages of fish, shellfish, crustaceans and marine animals) and "fish habitat" (spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes), as defined in the *Fisheries Act*, are used here. Shellfish aquaculture operations can result in an alteration to fish habitat by three basic mechanisms:

- the biodeposition, accumulation and remineralization of organic matter in shellfish feces and pseudofeces can alter benthic habitat as a result of changes in biogeochemical properties and biological community structure (flora and micro- to macrofauna);
- the removal of a significant fraction of the total natural seston in a bay to support the growth of cultured shellfish can alter the water column habitat (e.g. light environment and availability of food resources) for other marine organisms; and
- the translocation of organic matter remineralization from pelagic to benthic food webs and the excretion of ammonia by cultured shellfish can alter nutrient dynamics (e.g. recycling rates, retention of nutrients in coastal systems, nutrient ratios) and affect habitat and community structure.

Under some conditions all three mechanisms, in addition to potentially altering community structure, can also influence biological productivity, with potentially cascading effects on ecosystem structure and function. Examples of potential effects of aquaculture on productivity include:

- stimulation of primary productivity due to increased nutrient availability and cycling rate associated with the effects of shellfish grazing (Cranford et al., 2003);
- decline in the productivity of benthic infauna in the presence of toxic sulphides (Cranford et al., 2003); and
- increase in the productivity of demersal and macrobenthic predators attracted to feed on cultured species, fouling organisms, and on small polychaetes typically found in organically enriched sediments (ICES, 2005).

The strategy implemented by DFO Habitat Management for assessing aquaculture developments focuses on the potential for negative benthic effects on fish habitat (DFO, 2002). As a result, organic enrichment impacts from biodeposits are given a high priority in the selection of potential habitat indicators. However, to ensure that the principle of *no net loss of the productive capacity of fish habitat* is respected, potential productivity indicators also need to be assessed.

1.3.2. Cultured species

It is believed that certain culture methodologies and practices may have a greater potential for environmental impact than others (ICES, 2004). This primarily results from differences in the relative stocking density per unit area, which is directly related to the potential for the environment to absorb the impact. Based on current stocking densities and biomass, the longline culture of mussels is believed to have a relatively high potential for environmental impacts and is seen as a priority for the development of indicators. However, the general applicability of indicators to other aquacultured species is an important criterion for assessing indicator suitability.

1.3.3. Scales of potential impact

The issue of temporal and spatial scales of impact is an important consideration in the selection of any indicator, as the targeted effect should be measurable over a variety of scales. The indicator and measurement approach needs to be able to detect an impact over the actual temporal and spatial scale of the effect. Spatial scales extend from "local" (directly under and adjacent to the culture structure), "lease" (footprint), "bay" (the embayment or coastal management area), to "regional" (larger coastal areas with similar environmental conditions). Ideally, an indicator measurement taken at a specific sampling site should provide information on local impacts as well as impacts occurring over a much larger geographic scale. An important aspect related to spatial and temporal scales and indicator selection is the degree of variation in indicator values expected at aquaculture sites in Canada. Variation is directly related to the statistical design required to conduct effective and practical monitoring programs (e.g. identification of sample size) and the ability of the indicator to detect a known impact (i.e. statistical power).

Temporal scales of relevance to monitoring include embayment flushing times, and time-scales of physiological (e.g. food supply clearance time, phytoplankton turnover time) and biogeochemical (e.g. oxygen consumption and sulphide production rates) processes that all influence the capacity of the site to assimilate the perturbation. The above examples tend to have time scales measured from hours to days. However, longer time scales, such as the spring-neap tidal cycle, seasonal cycle and impact recovery time, all influence the degree of impact at aquaculture sites and need to be considered in the discussion of monitoring approaches.

1.3.4 Cost/benefit

Pragmatic considerations are paramount for identifying monitoring requirements for an industry that includes many small-scale operations. Low-cost measures of operational impacts are obviously preferable if they are able to contribute to the conservation and protection of fish habitat as effectively as more costly approaches. A balanced cost/benefit approach is also important when considering how the suite of available indicators (the "toolbox") is to be employed, because the "benefit" side of the equation is related to the predicted severity of the impact to ecosystem dynamics and fish habitat. Assessments of the benefits related to specific indicators also need to consider such criteria as the ease of data interpretation by managers, the availability of established and/or defensible theoretical reference points, and the ability to specifically identify aquaculture impacts in systems exposed to multiple stressors.

1.3.5 Demands of responsive management

For habitat/environmental management to be effective, the time-frame between data collection and the decision-making process needs to be as short as possible. Responsive and adaptive management approaches strive to implement mitigation measures quickly so that the marine habitat does not continue to deteriorate. Near real-time indicators therefore have a distinct advantage in such programs, whereas indicators that require considerable work to process samples and interpret data may be less desirable. Time lags greater than six months for scientists and managers to receive final results for an indicator can be considered undesirable.

2.0 BENTHIC HABITAT

2.1 Benthic performance indicators and thresholds (B.T. Hargrave)

2.1.1 Introduction

Suspended seston (phytoplankton, bacteria and resuspended sediment and flocculated detrital particles) is removed during feeding by both natural populations and cultured bivalves. The removal of particulate matter may be beneficial in preventing eutrophication in estuaries where agricultural runoff results in additions of dissolved nutrients that stimulate phytoplankton production (Cranford et al., 2003; Newell, 2004). On the other hand, production of feces by suspension feeders increases seston sedimentation and thereby contributes to benthic organic enrichment (Hatcher et al., 1994). Dead mussels can fall from socks and live animals may also be dislodged when they are cleaned of fouling organisms and harvested. If not rapidly consumed by predators, mussels reaching the bottom will usually die and their decomposing tissues provide an additional source of sediment organic enrichment.

2.1.2 Methods for assessing benthic organic enrichment due to shellfish aquaculture

Sedimentation and falloff, both highly site-specific measurements, are unlikely to be routine methods in a multi-site environmental monitoring program. More general techniques are required to assess benthic habitat impacts of potential organic enrichment in areas of intensive mussel aquaculture. A recent review of monitoring methods suitable for management of finfish aquaculture (Wildish et al., 2005) indicated a suite of recommended variables related to benthic impacts due to increased deposition of organic matter. Recommended methods depend on water depth, the nature and heterogeneity of benthic habitat types within a study region and the purpose of monitoring (e.g. evaluation of broad scale vs. local effects). Sediment grabs or cores cannot generally be used in areas with rocks or mixed cobble and gravel substrates but underwater photography and video recordings have been used successfully to determine benthic habitat conditions in these areas. One requirement is that criteria must be developed for qualitative or semi-quantitative image analysis (see Section 2.3). New methods using standardized scales of enrichment effects along an organic enrichment gradient may allow thresholds of effects to be determined from imaging methods (Wildish et al., 2004a). Sediment Profile Imaging (SPI) has been found to be a cost effective method for detecting changes in sediment structure and infauna communities in soft sediments (Wildish et al. 2005). Although it must be ground-truthed in each new location, changes in horizontal profile images of physical structures in surface sediments such as animal burrows, tubes and redox discontinuity depth allow quantitative measures such as the Benthic Habitat Quality Index to be derived to indicate the relative position of a site along the benthic organic enrichment gradient described qualitativey by Pearson and Rosenberg (Nilsson and Rosenberg 1997, 2000).

In areas where sediment grabs or cores can be used to obtain samples, a variety of sediment-related variables can be measured to quantify benthic organic enrichment (Table 2.1). The methods may be broadly grouped into geotechnical (sediment texture and porosity), geochemical (inorganic/organic chemical composition) and biological (microbial and fauna numbers/biomass and estimates of species diversity) techniques. Some of the methods, such as measurements of vertical gradients of dissolved nutrients in pore water, biomarkers or specific biochemical components such as amino and fatty acids, sedimentation measured with traps and benthic nutrient and gas flux require specialized equipment. The measurements can be time consuming and as many require either laboratory or *in situ* incubations, they are not suitable for monitoring purposes.

Observations of changes in numbers and biomass of benthic macrofauna represent the classic approach for determining effects of sediment organic enrichment (Pearson and Rosenberg, 1978; Wildish and Pohle, 2005). Subtle changes in environmental variables such as sedimentation and oxygen supply are often reflected in altered biomass or species composition of benthic macrofauna before they are detectable in sediment chemical properties. These observations can provide an early warning of changes in benthic community structure (Wildish and Pohle, 2005). They also provide a definitive measure of benthic community alterations that may be expressed as the percentage change (usually reduction associated with increased organic matter deposition) in species number, biomass and taxonomic diversity. For example, a HADD might be said to have occurred if environmental changes at an impacted site relative to reference or control locations in the same area result in a loss of a specified level of biodiversity. Without dedicated studies of the ecological impacts of potential community alterations, the effects on fish habitat are difficult to determine. While observations of benthic macrofauna remain the ultimate variable for monitoring environmental changes in benthic habitats, there are other problems in having these methods adopted as standard monitoring methods for Environmental Monitoring Program (EMP) purposes. Processing time for sorting and identification of macrofauna samples can also be long. Other approaches to environmental monitoring are required when information is needed in a short period of time (Wildish et al., 2001).

Of the variables listed in Table 2.1, geochemical properties such as water content (porosity) or grain size and selected chemical measures in surface sediments (organic matter (OM), redox potentials (Eh) and total sulfides (S: $S^{=}$; HS^{-}; and H_2S) have been found to be the most sensitive to organic enrichment (Hargrave et al., 1997). Horizontal gradients in these variables around salmon aquaculture sites were documented in the southwestern Bay of Fundy (SWNB) over a decade ago (Hargrave et al., 1997; Wildish et al., 2001). Since then, Eh (and pH in some jurisdictions), OM, and S have become widely used as standard variables to assess impacts of organic loading from finfish aquaculture (Brooks and Mahnken, 2003; Brooks et al., 2003; Holmer et al., 2005; Schaanning and Kupka-Hansen, 2005). These variables have been successfully applied in finfish EMP programs since they meet many of the criteria for methods to detect environmental

Table 2.1 Variables evaluated for their effectiveness to detect effects of sediment organic matter enrichment. Citations provide examples of methods applied to assess effects of shellfish aquaculture. Cranford et al. (2003) provide a more complete list of references. Asterisks indicate methods that have been applied to assess environmental effects of finfish aquaculture (Hargrave, 2003) but no publications evaluating effects due to shellfish culture using these variables have been identified.

Physical Variables
- water content (porosity) (Shaw 1998)
- texture (percent sand-silt-clay)*
- flocculation (grain size spectra and modal diameter)*

Chemical Variables
- total sulfides ($S^{=}$, HS^{-}, H_2S) (Shaw 1998, Cranford et al. 2003, Anderson et al. 2005)
- redox potentials (Eh) (Shaw, 1998; Cranford et al., 2003 ; Anderson et al., 2005)
- inorganic/organic matter (Shaw, 1998; Cranford et al., 2003; Anderson et al., 2005)
- organic carbon and nitrogen*
- dissolved oxygen (vertical profiles)*
- pore water gradients (O_2, CO_2, NH_4^{+}, PO_4^{3-}, $SO_4^{=}$)*
- trace metals (normalized for sediment grain size)*
- biomarkers (plant pigments- chlorophyll *a*/phaeopigments, fatty acids, hydrolyzable amino acids, C/N stable isotopes) (Hatcher et al., 1994)*

Biological Variables
- white sulphur (*Beggiatoa* spp.) bacterial mats (presence/absence, percent sediment cover) and benthic microbial response (Mirto et al., 2000)
- macrofauna (presence/absence) and taxonomic composition (taxa, family or species analysis level) used to derive measurements of species abundance and diversity (Shannon-Wiener Index) using measures of faunal numbers and/or biomass (Shaw 1998; Ragnarsson and Raffaelii, 1999; Chamberlain et al., 2001)
- trophic feeding group analysis (surface deposit to suspension feeder ratio, detritivores to carnivores, crustacean to non-crustaceans, Infaunal Trophic Index) (Shaw, 1998)
- benthic faunal community size spectral analysis (community analysis as body size composition)*

Process and Multivariate Measurements
- sedimentation rates (moored or bottom sediment traps) (Dahlback and Gunnarsson, 1981; Jaramillo et al., 1992; Hatcher et al., 1994; Chamberlain et al., 2001)
- benthic fluxes (O_2 consumption, CO_2 release, NH_4 release, PO_4 release) (Baudinet et al., 1990; Prins and Smaal, 1990; Barranguet et al., 1994; Hatcher et al., 1994)
- indices combining physical/chemical sediment variables (BHQ, RPD, BEI)*

changes discussed in Section 1.3. Some studies, however, have found that measurements of Eh and S either failed to detect enrichment effects or the expected relationship between the variables was affected by site-specific characteristics such as sediment texture, bottom heterogeneity, or quality of sediment organic matter. While an inverse relationship between Eh and S at salmon aquaculture sites in Newfoundland (NL) was similar to that described for SWNB, Eh-S potentials around mussel farms and reference sites in NL did not conform (Anderson et al., 2005). Heterogeneous bottom conditions around salmon farms in the Broughton Archipelago, British Columbia (BC) also led to variations in enrichment classifications in replicate samples from the same station (Sutherland et al., 2006, in press).

2.1.3 Benthic organic enrichment classification

Holmer et al. (2005) reviewed changes that occur in surface sediments and benthic microbial and faunal communities when hypoxic or permanent anoxic conditions are created, as discussed in the following section. This occurs when consumption of oxygen by bacteria in surface sediment layers exceed rates of oxygen supply by diffusion and advection. Restricted oxygen supply reduces aerobic microbial consumption of OM while anaerobic decomposition pathways are stimulated. Although some sulfide-tolerant macrofauna increase in abundance with moderate or even high levels of S accumulation, at some S concentration between 2 and 10 mM most macrofauna taxa are unable to survive (Brooks and Mahnken, 2003; Brooks et al., 2003; Wildish et al., 2005; Wildish and Pohle, 2005). Macrofauna are usually completely absent from anoxic sediments when S is >10 mM. OM preservation is therefore increased under hypoxic-anoxic conditions as numbers of macrofauna decrease. Bioturbation (sediment and pore water mixing), that tends to oxygenate sediments and increase the loss of OM, is also reduced as numbers of macrofauna decrease. Under these conditions, numbers and metabolic rates of sulphate-reducing bacteria increase and, in the absence of oxygen, S accumulates.

Microbial-macrofauna-geochemical variables described above are inter-related and change in a predictable way along an organic enrichment gradient. The gradients appear to be common for a variety of soft bottom marine benthic habitats (Wildish et al., 2001; Holmer et al., 2005). Where these geochemical indicators have been used in locations of shellfish (mussel) aquaculture, benthic enrichment effects have generally only been observed within or close to lease boundaries (Dahlback and Gunnarsson, 1981; Hatcher et al., 1994; Chamberlain et al., 2001; Cranford et al., 2003). As observed in studies around salmon aquaculture sites, changes in sediment geochemical variables indicating benthic enrichment associated with shellfish aquaculture are generally either within or restricted close to the edges of leases.

The oxic-hypoxic-anoxic classification, based on the inverse relationship between oxygen supply and S accumulation, is reflected in changes in microbial and macrofauna biota. Benthic oxygen demand and CO_2 and NH_4^+ release from sediments dramatically increase when S concentrations exceed 200 to 350 µM (Hargrave et al., 1993; Holmer et al., 2005) consistent with a transition from *normal* to *oxic* status as described in Wildish et al. (2001). As organic matter supply increases and oxygen availability becomes restricted, negative Eh potentials (0 to <-100 mV) in surface sediments indicate that sulfate reduction is a predominant metabolic process as reflected in S accumulation. The presence of different amounts of S can be detected by colour changes in sediments. Oxic conditions are usually indicated by sediments that are light brown or grey since S-metal complexes, if formed, are in low concentrations. Hypoxic deposits which contain variable amounts of complexed S are medium grey to dark brown. Highly reduced, anoxic sediments are dark grey to black due to FeS formation. While somewhat subjective, H_2S odour is detectable when S levels reach ~1000 µM. The smell of H_2S in sediment samples is used in some EMPs as a qualitatively indicator of sediment oxic-anoxic conditions. The odour of H_2S was detected at the transition from oxic to hypoxic sediments in the Broughton Archipelago (BC) area when S concentrations increased above 800 to 1200 µM (Brooks and Mahnken, 2003).

2.1.4 Evaluation of the generality of thresholds of benthic organic enrichment identified by total sulfides

Data sources

Geochemical measurements have been made in surface sediments collected from a variety of east and west coastal locations in Canada. Data from several of these studies were used to assess the generality and utility of Eh-S relationships for determining sediment oxic status in different areas. Observations were either part of provincial EMPs or were obtained during research projects in specific locations. Samples were usually taken at subtidal locations along transects or at individual stations around and adjacent to lease sites to determine sediment geochemical conditions. Intertidal sites are not represented in the data. Video surveys were used in most studies to indicate the degree of substrate heterogeneity and to verify that selected sampling stations were representative of bottom type at the site. Data were compiled in Excel spreadsheets using measurements from only surface samples (0-4 cm depth) with S values >50 µM. In many studies, Ag-AgS electrodes used for S measurements were calibrated using three standard S solutions (100, 1000 and 10000 µM) and therefore concentrations of S <100 µM can only be estimated by extrapolation. The rapid oxidation of S also makes it difficult to achieve accurate calibrations using 10 µM S standards. Exclusion of data <50 µM has no effect on the analysis since this is within the lower range of S that is characteristic of oxic sediments. In all cases, methods for measurements of Eh and S followed procedures described in Wildish et al. (1999), with triplicate samples from three grabs or cores taken at each station.

Station identification

In some locations, e.g. Broughton Archipelago area of BC, sediments up to 200 m away from lease boundaries may be influenced to some degree by particulate waste effluents from salmon aquaculture (Brooks and Mahnken, 2003; Brooks et al., 2003; Sutherland et al., 2006). Benthic organic enrichment gradients around salmon farms in more shallow water areas in SWNB extended over shorter (<50 m) distances (Hargrave et al., 1993). Since sedimentation of aquaculture-derived wastes occurs over variable distances around aquaculture sites, locations were only designated as reference if they were >200 m from lease boundaries. Data from mussel aquaculture lease and reference sites within one inlet in PEI and NS were compared with measurements in control (non-culture) inlets. Some aquaculture sites may not have been under production at the time of sampling but since information was not available to differentiate active vs. inactive sites, all licensed farm sites were identified as leases. In some studies, lease and associated reference sites were sampled simultaneously. These were analyzed as a single group since the aim was to determine if the oxic-hypoxic-anoxic classification system based on sedimentary Eh potentials and S concentrations was broadly applicable.

Eh-S Relationships

EMP data from surveys of salmon aquaculture farms in BC (2005), NS (2004) and New Brunswick (NB) (2004) (Fig. 2.1), from multi-inlet surveys in PEI embayments with and without mussel aquaculture (1997 and 2001), and lease and reference sites in Tracadie Bay (2003) PEI (Fig. 2.2) illustrate the expected inverse slope between Eh and S. A regression line (Eh = 473.4 - 65.95 Ln (S); r^2=0.67; n=80), derived from Eh-S measurements in surface sediments under and adjacent to salmon farms and at reference sites >500 m distant from farms in SWNB in 1994/95 (Hargrave et al., 1997; Wildish et al., 1999), is shown in each graph to provide a reference for comparative purposes. Since slope and intercept values of Eh-S regressions are highly dependent on ranges of S concentrations represented in the data or selected for analysis, separate regression lines were not calculated for individual data sets.

Despite high variance in all of the Eh-S plots (Figs. 2.1 and 2.2), the inverse relationship between the variables conforms to the regression derived from the 1994-95 SWNB data. Variations in sediment physical properties (grain size, porosity) were present to some degree at all sampling locations and this would be expected to create variability in relationships between Eh and S. Also, some of the variability may be attributable to methodological differences in sample collection since either sediment grabs or cores were used in individual studies. Eh and S were determined in surface sediments in one of two ways. When cores were available, the Pt electrode for Eh measurements was inserted horizontally into the surface 0-2 cm layer. After the Eh potential stabilized, a cut-off syringe was used to withdraw a subsample for S determinations. Eh and S were therefore determined for sediment from the same depth layer. Alternatively, when a grab was used, a cut-off syringe was inserted vertically into the sediment surface. Alternatively, when a grab was used, a cut-off syringe was inserted vertically into the sediment surface at a different location from where the syringe sample was taken. Cores were used to collect sediments during the NB 2004 EMP and for all measurements in PEI between 1997 and 2003. In general the variance in the Eh-S relationships in NB and PEI data sets appears to be lower than in plots of BC and NS EMP data where grabs were used for sampling. Increased variance in the Eh-S relationship in samples collected by grabs could reflect steep vertical gradients in the variables near the sediment surface where small differences in sampling depth for insertion of the Eh electrode and cut-off syringe would substantially affect redox conditions and S concentrations.

Another feature apparent in all Eh-S plots is that the variance of the relationship relative to the SWNB reference regression line increases as S concentrations decrease to <500 µM. This reflects the fact that oxic sediments are poorly “poised” with multiple redox couples that are not at equilibrium (Sigg, 2000). Mixed biological and chemical oxidation-reduction reactions occur in oxic sediments where metabolic products are accumulated from both aerobic and chemosynthetic anaerobic bacteria. Since there is no

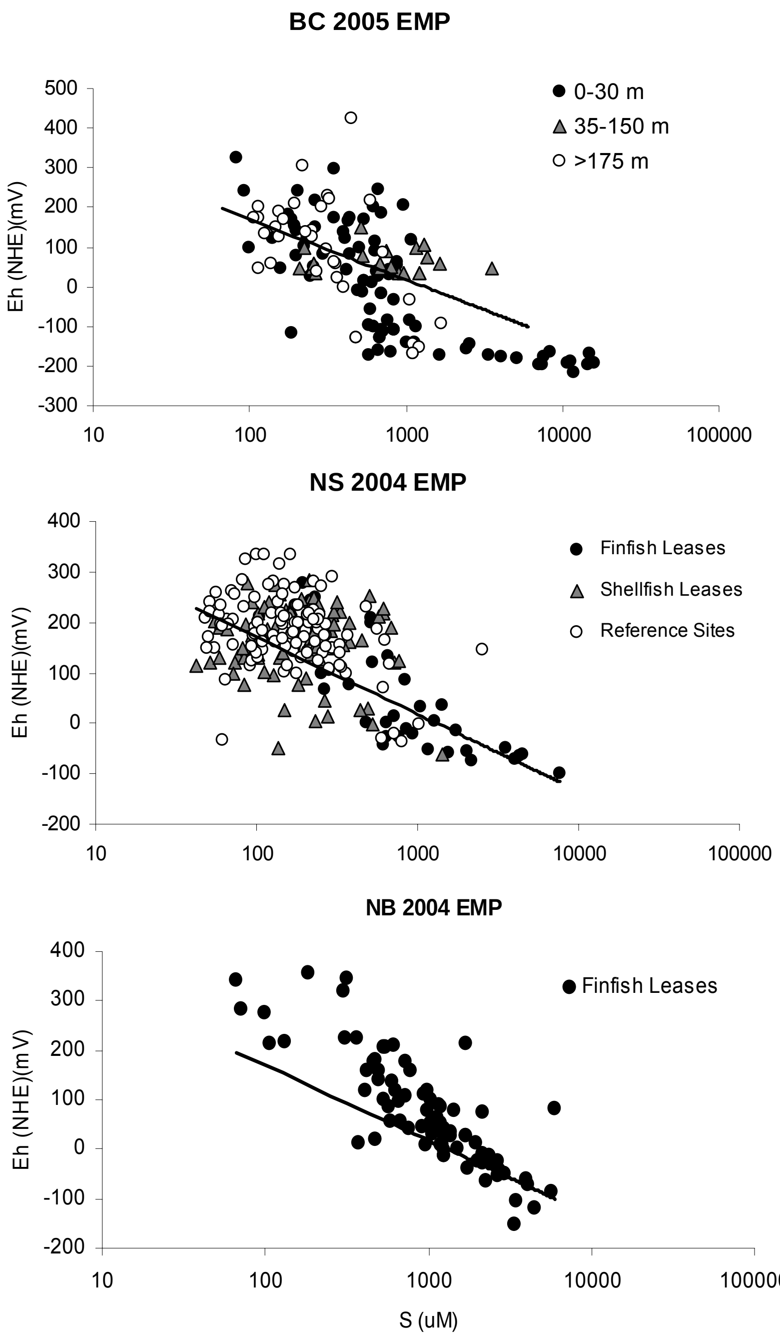


Figure 2.1. Comparison of Eh and S in surface (upper 2-4 cm) sediments collected as part of EMP sampling programs in BC (2005), NS (2004) and NB (2004). Data for these locations are summarized in Table 2.2. In some cases (BC and NS), sampling locations relative to aquaculture leases were recorded, allowing data to be classified with respect to distance from farm boundaries or reference sites. The diagonal line represents the regression line Eh = 474 - 65.95 (Ln x) in Wildish et al. (1999), derived from data for samples collected under net pens and >500 m away from salmon aquaculture lease sites.

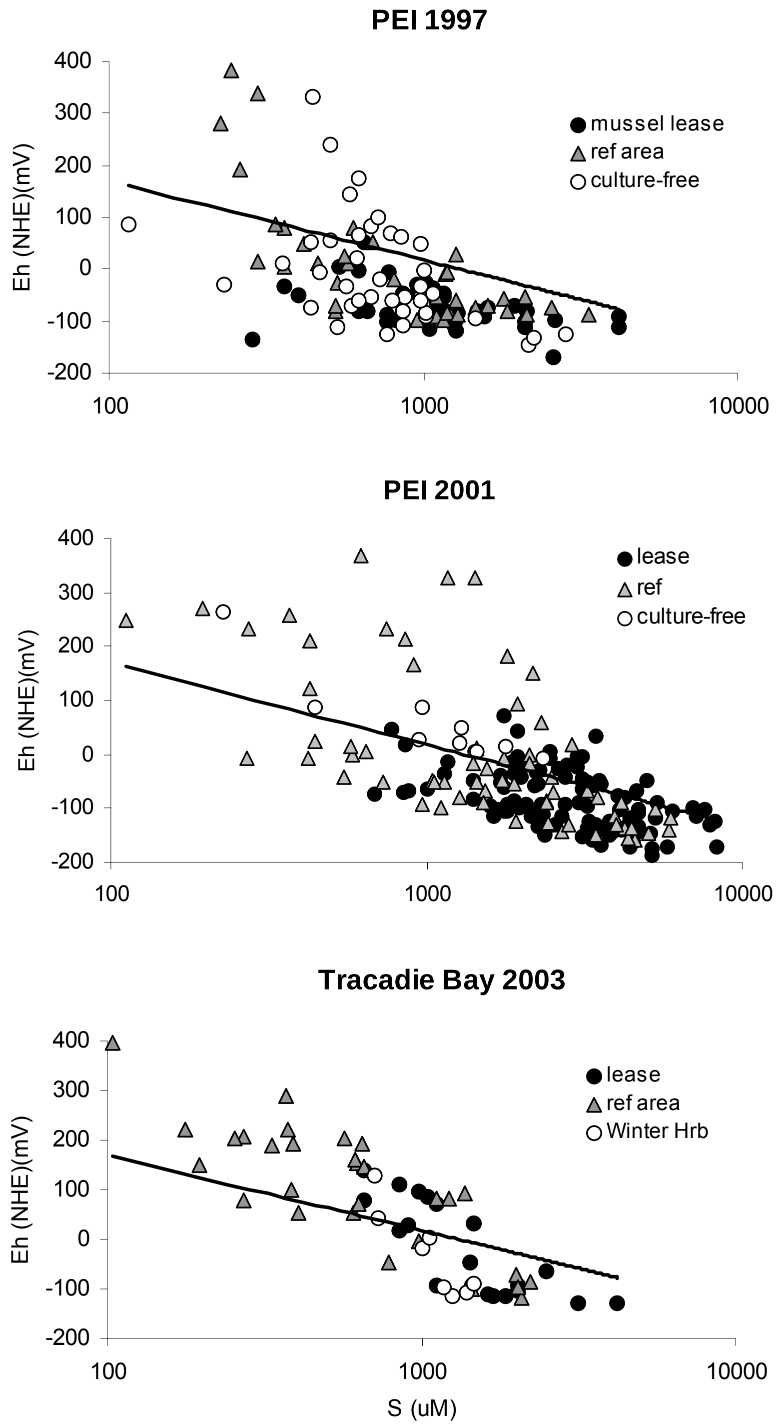


Figure 2.2. Comparison of Eh and S in surface (upper 2 cm) sediments collected during multi-inlet surveys in 1997 in PEI (Shaw, 1998) and repeated in 2001 (B. Hargrave, unpublished data) and a more detailed survey of Tracadie Bay, PEI, in 2003 (B. Hargrave, unpublished data). Data for these locations are summarized in Table 2.2. The diagonal regression line is described in Fig. 2.1.

single redox couple responsible for creating reducing conditions, Eh and S are poorly correlated in oxic sediments. Low S concentrations are also highly variable in these deposits since S compounds are rapidly oxidized in the presence of oxygen and subject to effects of sediment mixing and oxidation through bioturbation. In more reducing sediments (S concentrations >1000 µM), the variance in the Eh-S relationship decreases. These sediments are dominated by sulfate reducing bacteria. Negative (<0 mV) Eh potentials reflect the importance of sulfate reduction as the primary metabolic reaction determining the formation of reduced sulfides. The absence, or limited availability, of oxygen also allows reduced S to accumulate.

An important factor to be considered when comparing Eh-S data from different areas is that with extended use, the Pt surface of Eh electrodes can become 'poisoned' with sulfide and oxide coatings (Wildish et al., 2004b). Cleaning of electrodes can sometimes, but not always, restore electrode response. When such coatings are formed, the exchange of electrons across the Pt surface is reduced and, for a given S concentration, lower potentials are measured than would be expected.

Assessing sulfide thresholds for identifying oxic-hypoxic-anoxic sediments

Non-parametric K-means clustering (Euclidean distance) with *z*-score standardized Eh and S values using Systat© (SPSS Ver. 10) (Wilkinson, 2000) was used to identify groups of stations based on 13 data sets representing salmon, mussel and oyster aquaculture areas in five provinces (BC, NB, NS, PEI and NL) (Table 2.2). The method divides a set of objects with measured characteristics into a selected number of groups by maximizing between-cluster variation relative to within-cluster variation. Cluster identification is equivalent to a one-way analysis of variance where groups are unknown. Standardization is required to normalize variable scales.

K-means separation of observations, based on Eh and S, corresponded to ranges of S concentrations characteristic of Oxic-Hypoxic-Anoxic conditions described in earlier studies (Wildish et al., 2001; Holmer et al., 2005; Wildish and Pohle, 2005) (Table 2.2). In some data sets, a second analysis was required using only data in the Oxic A and B ranges to specify K-means separation into two groups. The ranges of S concentrations characteristic of Oxic A, Oxic B, Hypoxic A, Hypoxic B and Anoxic sediments in all locations were summarized to derive descriptive statistics for each organic enrichment group (Table 2.3). Rounded mean and median values are generally consistent with previous descriptions of oxic-anoxic gradients based on Eh and S relationships in sediments near and adjacent to finfish aquaculture sites (Wildish et al., 1999; 2001; Brooks et al., 2003; Holmer et al., 2005; Wildish and Pohle, 2005). The maximum S threshold level for Oxic B conditions (1500 µM) is slightly higher than the value previously identified as a maximum S concentration for oxic deposits and the range of Eh potentials characteristic of this enrichment class is slightly broader (+100 to -50 mV) than previously proposed (S of 1300 µM and Eh of +100 to 0 mV). The upper boundary for fully anoxic sediments remains at S>6000 µM with associated Eh potentials <-150 mV compared with the previous Eh threshold of <-100 mV characteristic of grossly polluted sediments.

Table 2.2. Ranges (minima-maxima) of S (µM) and number of samples in sediment organic enrichment groups identified using K-means clustering (Wilkinson, 2000) based on S and Eh (mV) potentials in surface (0-4 cm) sediments in 13 different locations. Enrichment classes are defined in Wildish et al. (2001). Samples with S<50 µM are excluded from the analysis as explained in the text. Provinces, year(s), locations and data sources (literature citation or personal communication of unpublished data) indicate where studies were conducted. EMP indicates provincial environmental monitoring programs. Site type represents finfish (salmon) (FF), mussel (M) or oyster (O) leases (L), reference locations (R) and non-culture inlets or bays (NC). Numbers in parentheses associated with site type indicate distances (m) from finfish lease boundaries. NP indicates that no samples in this organic enrichment class were present.

Location	Data Source	Site Type	Oxic A	Oxic B	Hypoxic A	Hypoxic B	Anoxic
BC EMP	Taekema et al.	FFL (0-30)	83-962 (19)	99-1090 (31)	199-3350 (28)	4080-8410 (7)	10900-15900 (6)
(2005)	(2006)	FFL (35-150)	211-797 (17)	894-1660 (12)	3570	NP	NP
		FFL (>175)	106-716 (27)	487-1680 (9)	NP	NP	NP
NS EMP	T. Balch	FFL	NS	104-839 (15)	484-2171 (16)	3501-4462 (4)	7626 (1)
(2004)	(NSDAF)	ML	50-378 (79)	241-774 (16)	1428	NP	NP
		R	50-478 (109)	561-1012 (8)	NP	NP	NP
NB EMP	E. Parker	FFL	66-319 (6)	106-1684 (17)	368-2167 (47)	2147-5980 (19)	NP
(2004)	(NBDELG)						
PEI 1997	Shaw	ML	539-768 (4)	360-1163 (10)	619-2140 (20)	2594-4188 (5)	NP
(multi-inlet)	(1998)	R	227-528 (7)	244-1458 (23)	1600-3343 (7)	NP	NP
		NC	115-975 (13)	233-1457 (24)	2154-2857 (3)	NP	NP
PEI 1998	Wildish et al.	ML	63-780 (13)	990-2800 (9)	1200-2900 (10)	5700	NP
(Tracadie Bay)	(1999)[1]	R	120-150 (2)	160-870 (7)	150-1900 (9)	3500-4500 (2)	NP
PEI 2001	B. Hargrave	ML	227	441-1954 (13)	1143-3588 (29)	3236-5773 (65)	6028-8319 (7)
(multi-inlet)		R	112-2176 (14)	271-2310 (27)	1928-3477 (12)	3494-5967 (11)	NP
		NC	227	441-966 (3)	1275-2347 (6)	NP	NP

Table 2.2. Continued.

Location	Data Source	Site Type	Oxic A	Oxic B	Hypoxic A	Hypoxic B	Anoxic
PEI 2003 (Tracadie Bay)	B. Hargrave	ML	NP	655-1470 (9)	1120-2480 (9)	3140-4200 (2)	NP
		R	104-653 (14)	270-1370 (10)	1450-2220 (5)	NS	NP
Broughton Archipelago (BC) (2000-2003)[2]	T. Sutherland	FFL+R	59-721 (20)	74-852 (15)	800-3600 (6)	5830-6380 (2)	NP
Lameque Hrb (NB)(2000-2005)	S. Courtenay	ML+R	110-1260 (9)	420-1465 (26)	1260-2845 (21)	3160-6100 (5)	6340-9100 (4)
Newfoundland Bays (2001-2005)[3]	R. Anderson	FFL+ML +R	109-1040 (12)	67- 676 (12)	1078-2424 (5)	2971-4976 (3)	NP
St. Ann's Hrb (NS)(2000-2004)	DFO, NSDAF	ML	69-391 (33)	111-1270 (32)	NP	NP	NP
	J. Grant	R	51-296 (32)	501-705 (4)	NP	NP	NP
St-Simon Bay (NB) (2002-03)	A. Mallet	OL (tables)	134-819 (23)	330-1640 (30)	NP	NP	NP
		OL (flo-bags)	74-993 (44)	250-2120 (33)	NP	NP	NP
		R	155-1380 (62)	206-1990 (45)	NP	NP	NP
SWNB (1994/95)	Hargrave et al. (1997)	FFL (0)	NP	180-2150 (31)	710-4800 (37)	750-4800 (43)	6400-14000 (13)
		R	51-280 (17)	50-700 (16)	NP	NP	NP

[1]S concentrations reported in Appendix 2 of Wildish et al. (1999) were divided by 10 to correct for a presumed calibration error
[2]includes data from Sir Edmund Bay reported in Sutherland et al. (2005)
[3]includes data from 2001-2003 reported in Anderson et al. (2005)

Table 2.3 Descriptive statistics summarizing results of K-means cluster analysis based on S data presented in Table 2.2. Mean values represent the minimum to maximum ranges of S (µM) and Eh (mV) representative of each sediment organic enrichment group. n is the number of data sets used to derive associated values. Threshold values for S and Eh in each enrichment group are based on rounded mean and median values.

Enrichment Group	Oxic A		Oxic B		Hypoxic A		Hypoxic B		Anoxic	
					S (µM)					
Mean	121	766	324	1412	1032	2859	3200	5478	7417	11830
SE	22	97	47	108	139	183	343	352	1164	1850
Median	105	745	250	1457	1120	2845	3198	5375	6370	11550
Min	50	150	50	676	150	1900	750	4188	6028	8319
Max	539	2176	990	2800	2154	4800	5830	8410	10900	15900
n	22		27		17		12		4	
Threshold (S µM)	**<750**		**750-1500**		**1500-3000**		**3000-6000**		**>6000**	
					Eh (mV)					
Mean	314	78	114	-53	-7	-105	-74	-177	-124	-183
SE	31	22	23	15	18	16	29	19	31	24
Median	355	81	97	-66	-5	-116	-99	-157	-135	-194
Min	36	-118	-70	-170	-117	-261	-249	-362	-179	-225
Max	504	267	399	136	120	-7	83	-109	-47	-120
n	21		26		16		12		4	
Threshold (Eh mV)	**>100**		**100 to -50**		**-50 to -100**		**-100 to -150**		**<-150**	

The identification of Hypoxic A and B sediment organic enrichment classes (Table 2.3) indicates that a transition in sediment types occurs within the Hypoxic sediment category at approximately 3000 µM S. Identification of this threshold level is useful for indicating when mitigation and remediation efforts might be required if changes occur in sediment oxic status as a result of aquaculture development. This is discussed further in Section 6.4. The previous range specified for Hypoxic conditions (1300-6000 µM S) was broad and the identification of an intermediate value (3000 µM S) may be useful for management and regulatory purposes.

While not the primary focus of the K-means cluster analysis, Table 2.2 can be used to examine Eh-S data from within and adjacent to lease sites to test the sensitivity of the approach to detect differences in sediment oxic status. For example, data from SWNB in 1994/95 clearly show the well-known impact of increased organic matter sedimentation immediately under salmon aquaculture pens. S concentrations in samples collected under net pens (0 m distance) ranged from 180 to 14000 µM with four organic enrichment categories (Oxic B to Anoxic) represented. Reference sites >500 m distant from pens in SWNB, on the other hand, had only Oxic A and B enrichment categories present with no Hypoxic or Anoxic sediments. A similar pattern with distance from salmon lease sites appears in the BC EMP 2005 and NS EMP 2004 data sets where sediment was collected adjacent to and at various distances from salmon pens. No Hypoxic or Anoxic sediments were present in the BC EMP 2005 data set when distances from net pens were >175 m or in NS at reference sites. Only one Hypoxic A site occurred at distances of 35 to 150 m away from salmon farm sites in BC and at mussel lease sites in NS where all samples were otherwise in the Oxic A and B categories. However, in both the BC and NS EMP data, all organic enrichment classes, including Hypoxic and Anoxic sediments, were represented when samples came from close to (0-30 m in BC EMP data) salmon farm sites.

The analysis shows that all sedimentary organic enrichment classes (Oxic to Anoxic) also occurred in multiple PEI inlets sampled in 1997 and 2001 (Table 2.2). Oxic to Hypoxic conditions occurred in surface sediments from Tracadie Bay in 1998 and 2003. In general, S concentrations at mussel lease and reference sites in the same PEI inlet had overlapping ranges but there was a trend towards lower S concentrations in reference locations. A notable change appears to have occurred in Tracadie Bay between Shaw's study in 1997 and DFO sampling at the same stations in 2001. Hypoxic B sediment were not present at any reference sites sampled in 1997, whereas 11 of 64 (17%) of the reference sites sampled in 2001 were in this sediment enrichment category. The effects of high levels of organic matter supply from multiple sources in all PEI inlets not associated with mussel aquaculture is shown by the presence of Hypoxic A sediments in sediments from non-culture inlets. It is notable, however, that in Tracadie Bay, and all other inlets sampled between 1997 and 2003, Hypoxic B and Anoxic sediments indicative of higher levels of benthic organic enrichment were only observed within mussel lease boundaries. The conclusion that increased OM sedimentation results in S accumulation in sediments in PEI inlets where intensive mussel aquaculture development has occurred, is supported by a detailed analysis of results from Tracadie Bay.

The K-means analysis also showed that, when compared to reference locations, sediment organic enrichment effects at mussel aquaculture sites in NS and NB are less extensive than at finfish farm sites. With the exception of one Hypoxic A site in the NS EMP 2004 data, only Oxic A and B enrichment categories occurred at shellfish lease and reference sites in NS EMP 2004 (mussels), St. Ann's Harbour NS (mussels) and St-Simon Bay 2002 NB (oysters). Oxic A and B sediments predominated in these locations in contrast to sediment conditions at salmon lease sites sampled during the NS EMP 2004, where some samples showed the presence of Hypoxic B and Anoxic sediments. The recognition that only oxic sediments occur at farm and reference locations, as shown in data from the 2004 NS EMP, St. Ann's Harbour NS, and St-Simon Bay NB (Table 2.2), may be relevant to recommendations for appropriate levels of environmental monitoring discussed in Section 6.4.

An example of detection of sediment enrichment effects due to multiple sources of organic matter is provided by data from Lameque (Shippagan) Harbour NB (Table 2.2). Several sources of industrial effluent, including discharges from a seafood processing plant and municipal wastewater, enter the harbour. A mussel aquaculture site, as well as naturally occurring offshore sediment transport along a central channel dredged to facilitate shipping, may also increase sedimentation of organically rich material in the relatively deeper, central area of the harbour. All five sediment organic enrichment categories are represented in surface sediments from Lameque Harbour. In samples (n=6) collected in February 2005 adjacent to a mussel culture site, S concentrations (965-1,620 µM S) were characteristic of Oxic B-Hypoxic A enrichment categories. Values were slightly higher than those in shallower water near the seafood plant effluent discharge (416-1,465 µM S), but were significantly lower ($p<0.05$) than concentrations in sediments from deeper water within the harbour channel (1,380-2,845 µM S).

2.1.5 Conclusions

1. When oxygen supply is restricted or reduced, and organic loading is sufficient to cause anaerobic conditions at the sediment-water interface, the progressive development of anoxic conditions can be detected by Eh and S measurements.
2. Eh and S relationships in different areas have been used to identify the thresholds of S concentration characteristic of Oxic A, Oxic B, Hypoxic A, Hypoxic B and Anoxic sediments. Additional data is required to extend the comparison to intertidal sediments. Tidal currents and wave action causing resuspension and transport in intertidal areas should in general result in oxic sedimentary conditions. Exceptions may occur if excessive organic matter supply to sediments due to the presence of cultured organisms or physical structures in the intertidal zone is sufficiently high to cause the formation of hypoxic or anoxic conditions.
3. S and Eh in sediments from 13 locations on Canada's east and west coasts are inversely correlated in a similar manner at both finfish and shellfish aquaculture sites.
4. Changes in sediment oxic status in inlets where mussel aquaculture has been developed in PEI and NS appear to be less than changes observed around some salmon aquaculture farms where high rates of waste feed and fecal matter sedimentation have led to the formation of Hypoxic and Anoxic conditions in surface sediments.
5. Since hydrographic and physical conditions (water depth, currents, bottom substrate type) determine particulate matter deposition at any given location, organic matter accumulation in or on the bottom and resulting changes in benthic oxic status due to aquaculture, can be highly variable within a small area.
6. Eh-S observations that have been shown to represent a cost-effective approach to determining levels of sediment organic enrichment associated with finfish aquaculture can also be applied to assess the oxic status of marine deposits associated with shellfish aquaculture.

2.2 Benthic Communities (P. Archambault and M.D. Callier)

2.2.1 Introduction

Many studies have shown that benthic communities may be affected by the addition of high levels of nutrient and organic matter to the environment (Pearson and Rosenberg, 1978; Mattsson and Lindén, 1983; Brown et al., 1987; Diaz and Rosenberg, 1995). Pearson and Rosenberg (1978) proposed a model of changes in the benthic community with increasing organic enrichment. This model could be applied, to some extent, near a source of organic enrichment caused by shellfish farming. It is well recognised that with an increase in organic loading, spatially or temporally, the macrobenthic community is expected to exhibit (Weston, 1990):

1) a decrease in species richness and an increase in the total number of individuals as a result of the high densities of a few opportunistic species;
2) a general reduction in biomass, although there may be an increase in biomass corresponding to a dense assemblage of opportunist species;
3) a decrease in body size of the average species or individual;
4) a shallowing of that portion of the sediment column occupied by infauna; and
5) a shift in the relative dominance of trophic groups.

However, these putative changes are more difficult to identify in benthic assemblages distant from aquaculture sites. Many indices have been used to detect changes in microbial (Mirto et al., 2000; Danovaro et al., 2004b), meiofauna (between 37 µm to 1mm) (Mirto et al., 2000; Danovaro et al., 2004b) and in macrofauna (>1 mm or 0.5 mm) (Mattsson and Lindén, 1983; Kaspar et al., 1985; Grant et al., 1995; Hartstein and Rowden, 2004) related to shellfish farming. This section reviews the existing knowledge and research needed to identify indices that are sufficiently sensitive to detect changes in benthic communities due to shellfish farming over short and/or long spatio-temporal scales. Specifically, appropriate indices and thresholds will be highlighted for managing the impacts of shellfish farming.

The different objectives and indicator groups (microbial to macrofauna) focused on in impact studies and the different meanings of the terms “indices” or “indicators” used by scientists and managers, have led to an excess of criteria that have been proposed to characterize the “ideal” indices. Many of these criteria represent limits of biological extremes (see Table 2.4; modified from Jones and Kaly, 1996). Studies using species as indices are often based on the same biological parameters, but at the opposite ends of the scale (Table 2.4). For example, one species can become abundant and another rare under the same disturbance. This has led to different results in many studies and has only increased the complexity for decision making. To avoid this problem, the question asked must be “clear” (Jones and Kaly, 1996; Downes et al., 2002). For example, has the abundance of *Capitella* changed, or what is the number of species? The indices that are used will change to some extent at one or many spatial and temporal scales. For this reason, again, the question of what indices should be studied needs to be identified *a priori*. It is important to keep in mind, as mentioned by Cullen (1990), that managementoriented questions should be addressed in a Popperian scientific framework. Cullen (1990) argues that ignorance of the appropriate spatial and temporal scales has limited the utility of scientific studies to decision makers.

Table 2.4. Double-ended criteria for choosing indicator organisms (modified from Jones and Kaly, 1996).

Stress		
	tolerant	susceptible
Abundance		
	common	rare
Distribution		
	cosmopolitan	localized
Population stability		
	stable	unstable
Life history		
	long-lived	short-lived
Habitat		
	specialist	generalist
Mobility		
	sessile	mobile

The identification of a threshold for a single variable needs to be decided collectively, by relevant parties (decision-makers, environmental groups, scientists, etc.; Downes et al., 2002). The choice of a threshold for a particular variable should not be considered lightly and any modification of this threshold necessitates a reconsideration of what the relevant parties have agreed to be important for a specific activity, in this case shellfish farming.

2.2.2 Indices

The choice of indices is a critical decision and should not be made arbitrarily. Many studies have used sediment chemistry indices to identify changes related to aquaculture activities because these parameters are easily measured and are well-correlated with the latter. However, Weston (1990) and Edgar et al. (2005) found that biotic indices were more accurate and more sensitive to aquaculture activities at different spatial scales than sediment characteristics. The choice of indices centers on the question asked. For aquaculture-related activities, we are mainly concerned with habitat productivity (i.e. biological indices). The question asked should be precise and might be, for example: how could shellfish farming change the density of lobsters in the next *n* years? A poor question would be: how could shellfish farming change the "health" of an ecosystem? This is difficult to answer and will result in considerable confusion, because "health" is a term that does not have a universally accepted definition.

The question asked should be relevant. The soundness of a question is related to the quality of the indicator. First, a good indicator should be *causally or strongly associated* with shellfish farming. For example, species richness is often used as an indicator of anthropogenic impact because it is a common belief that it will decline with human activities: this is not always the case (Drouin et al., 2006). Profound changes in community composition could occur without any alteration in the overall number of species (Keough and Quinn, 1991; Drouin et al., 2006) but could be ecologically important for the productivity of an ecosystem. The second quality is the *efficiency* of an indicator. The *ecological and societal significance* of an indicator is also an important factor to consider (see Fairweather, 1999). Indices that have all these characteristics are rare. The choice of an indicator should be based on the careful examination of each indicator and its efficiency, in particular under the same conditions (sampling design, site, spatial and temporal scales, etc.). To our knowledge, no studies have investigated a set of biological, sediment and/or water chemistry indices under similar conditions to identify which indices are best suited for shellfish aquaculture. For fish farming activities, Edgar et al. (2005) suggested that biotic indices were more sensitive than abiotic indices. Of several biotic indices tested, only the redox potential at a 40 mm depth was found to be sufficiently sensitive to detect change.

Here, we summarize the most common benthic community indices used to identify the influence of shellfish aquaculture. These indices are classified into three major categories, based on the definition of Washington (1984). The first category consists of “diversity indices”. There are a plethora of diversity indices (see Magurran, 2004) but the focus will be on the number of species, the Shannon-Wiener diversity (H’), the evenness (J’) and the species richness (d). The second category, the “biotic indices”, corresponds to the total abundance, the biomass and the density of indicative species. The last category, the “similarity indices”, measures the similarity of the structure of two communities. Table 2.5 summarizes thirteen studies that have used these various indices to examine the influence of shellfish aquaculture on benthic communities.

Diversity indices

Most studies on the influence of shellfish farming used biodiversity indices to identify the intensity of changes. Most studies that included measurements of community variables showed significant decreases in the total number of species (Mattsson and Lindén, 1983; Kaspar et al., 1985; Christensen et al., 2003; Callier et al., 2005), Shannon-Wiener diversity (H’) (Mattsson and Lindén, 1983; Kaspar et al., 1985; Stenton-Dozey et al., 1999; Chamberlain et al., 2001), and the species richness (d) (Stenton-Dozey et al., 2001) of macrobenthic fauna under mussel cultures. However, some studies observed the opposite response (higher total biomass, higher diversity H’ and lower dominance) at mussel farms (Grant et al., 1995). Grant et al. (1995) compared the effectiveness of various methods used for assessing macrofaunal community structure. At their control site, the diversity was lower due to the dominance of one species (*Nephtys*), which is not the classical response expected.

Biotic indices

1) Indicator species. Highly polluted marine sediments are generally dominated by a few opportunistic macrofaunal species, such as *Capitella* sp. This small polychaete is tolerant of high organic enrichment and low oxygen conditions. Other deposit-feeding polychaete taxa, such as *Malacoceros* and *Ophryothrocha* sp., have been observed in enriched sites in northern Europe (Pearson and Black, 2001). Large carnivorous nematode worms also dominate highly enriched areas and phyla such as Mollusca and Echinodermata are completely excluded. In moderately enriched sediments in boreal latitudes, Capitellid, Spionid and Cirratulid families generally dominate, with small bivalves of the Tellinid and Erinacean families. Under normal conditions, these intermediate populations are gradually replaced by a diverse but less dense community (Pearson and Black, 2001), which includes molluscs, echinoderms, crustaceans and polychaetes. Mattson and Linden (1983) showed that the dominant species from the original fauna *Nucula nitidosa, Echinocardium cordatum* and *Ophiura* spp. were replaced by three polychaetes *C. capitata*, *Scolelepis fuliginosa* and *Microphthalmus sczelkowii* due to mussel culture biodeposition. Christensen et al. (2004) have also recorded enhancement of small surface-deposit-feeding polychaetes, *D. incerta, C. capitata and Prionospio* spp. at a mussel farm site. This appears to have crucial implications for oxygen penetration because of the lower bioturbation capacity of these small invertebrates.

2) Trophic indices. The general model (Pearson and Rosenberg, 1978) indicates that in highly organically enriched areas, benthic communities are dominated by deposit-feeders. Numerous studies observed a shift in macrofaunal community structure from suspension feeders to deposit feeders and scavenging gastropods (Grant et al., 1995; Stenton-Dozey et al., 1999). The absence of suspension feeders may be a good indicator of perturbation because organic debris has a smothering impact preventing suspension feeders from thriving (Kaspar et al., 1985).

3) Index of Biotic Integrity (IBI). To our knowledge, no studies on the effects of shellfish farming have used Integrated Biotic indices. IBI are limited to the geographical areas where the tolerance list has been compiled (Washington, 1984).

Similarity indices

Benthic community structure. Comparison of community structure with similarity analysis is a sensitive indicator since it is possible to determine differences among sites even at low organic enrichment (Warwick and Clarke, 1991; Downes et al., 2002; Drouin et al., 2006). Along a gradient of increasing organic enrichment, a continuous faunal succession occurs. Pearson and Rosenberg (1978) described four stages of community succession. Specific organisms are associated with particular levels of enrichment. Wildish et al. (1999; 2001) outlined the different levels that could be used to determine the effects of fish farming.

Consistent trends in Table 2.5 show that reliable indices are similarity indices (benthic community structure) and biotic indices (indicator species and trophic group). More specifically, changes in benthic community structure have been observed between control and farm sites, and along transects leading away from farms. Table 2.5 illustrates shows the effectiveness of monitoring for opportunistic, deposit feeding and scavenger species which tend to increase in number under shellfish farms. The opportunistic

Table 2.5. Review of studies on the influence of shellfish farms on benthic communities. Summary of the indices used: A- Diversity indices (univariate), 1-Total number of species, 2-Total abundance, 3-Total biomass, 4-Shannon-Wiener diversity indices (H'), 5-Species richness (d), 6-Evenness (J'); B-Biotic indices, 7-Indicative species, 8-Dominant species, 9-Trophic group; C-Similarity indices, 10-Community structure.

Author	Diversity	Biotic	Similarity	Max distance from control site	Protocol	Observations at farm site(s)compared to control site(s) Or in direction of farm site along transect
(Crawford et al., 2003) Tasmania- 3 farms	1-2-4	8	10	35-100m	3 transects 7-9 stations	NS
(Chamberlain et al., 2001) Ireland- 2 farms	1-4	8	10	40-60m	1 transect 3-4 stations	Farm-1 • Dominant species density increase • H': no significant difference • MDS: no clear difference Farm-2 • Deposit-feeders dominance • Opportunistic dominance • H' decrease • Difference in community structure
(Christensen et al., 2003) New Zealand	1-2	7	10	250m	Transect 3 stations: 0, 5, 250 m 1 reference bay	• Number of species decrease • Bioirrigating species decrease • Opportunistic species abundance increase
(Danovaro et al., 2004a) Mediterranean	Meio 1-2		10	600m	3 control sites 3 mussel sites	• Variations between sampling period > between sites • Bacterial abundance higher in autumn • No difference in meiofaunal abundance
(Hartstein and Rowden, 2004) New Zealand- 3 farms		7	10	200m	4 control sites 4 mussel sites	Farm-1 and 2 • Variation between site > variation between sampling period • Dominance of opportunistic polychaetes • Disappearance of the ophiurids *Amphiur* spp. Farm-3 • No difference in community structure (stronger tidal current)

Table 2.5. Continued

(Grant et al., 1995) Canada	1-2-3-4	7	10	30m	1 control site 1 mussel site	• Lower abundance • Higher H’ • Similar species composition • Difference in abundance of dominant species
(Kaspar et al., 1985) New Zealand	1-3		10	1km	1 control site 1 mussel site	• Biomass did not show difference because of patchiness • Lower diversity • Dominance of Polychaetes
(Stenton-Dozey et al., 1999) South Africa	2-3-4-5-6	7-9	10-11	750m	3 transects 3-4 sites per transect + 9 mussel sites	• Species richness (d) decreased along all transect • H’ only decreased along one transect • J’ remain constant. • Deposit-feeder dominated at all sites. • Carnivores were the second dominant group at mussel farm, while suspension feeders were dominant group at the reference site
(Yokoyama, 2002) Japan	1-2-3-4-5-6	8		6 km	1 control site 1 mussel site	• Lower abundance, H’ and d • More unstable community
(Mattsson and Lindén, 1983) Sweden	1-2-3-4	7	10	50m	1 transect 5 sites	• Peak of opportunistic *Capitella* in April • H’ decrease • Fluctuation of abundance and biomass • Opportunistic species increase
(Mirto et al., 2000) Mediterranean	Meiofauna: 2-		Meiofauna: 10	1km	1 control site 1 mussel site temporal replication	• Lower meiofaunal density (turbellarian, ostracod, kinorynch) • Increase in bacterial density
(D'Amours and Archambault, 2005) Canada- 5 farms	1-2		10	2 km	Transect 0, 50, 100, 500, 2000 m	• No difference in epibenthic megafauna composition • Higher total abundance
(Callier et al., 2005) Canada- 2 farms	1-2-3	7	10-11	300m	Transect 7 sites (0, 3, 6, 9, 15, 30, 300m)	Farm-1 • Lower abundance • Lower number of species • Biomass (Fig. 2.3) • Difference in benthic community structure (Fig. 2.4) Farm-2 • NS

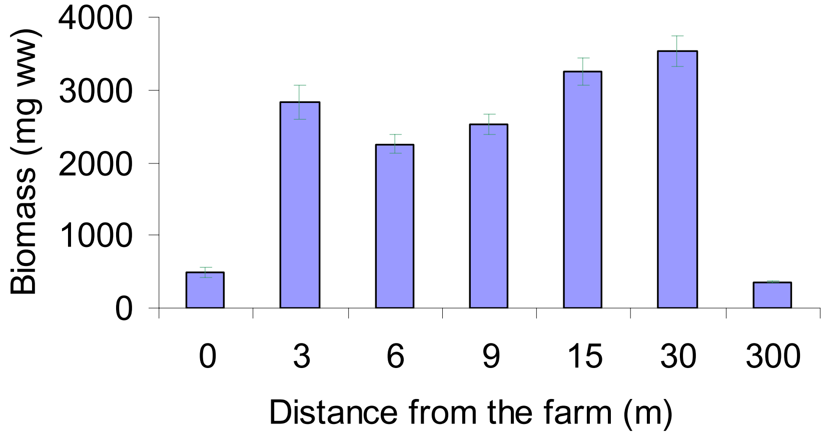


Figure 2.3. Mean infaunal biomass (±SE) along transects placed perpendicular to the last mussel lines of a mussel farm located in the Magdalen Islands, Quebec in 2004 (Havre-aux-Maisons lagoon) (Callier et al. 2005).

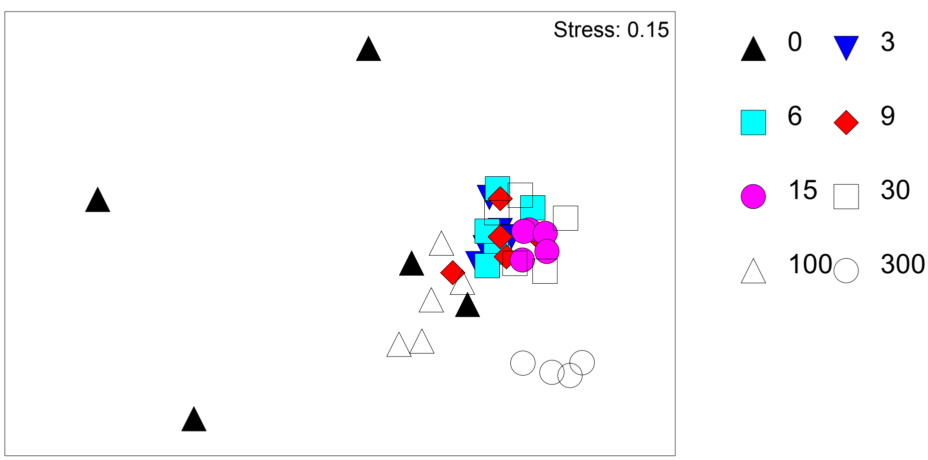


Figure 2.4. Non-metric multi-dimensionally scaling ordinations of infauna assemblages in sediment cores (n= 5) for 8 distances (0, 3, 6, 9, 15, 30, 100 and 300 m) along a transect perpendicular to a mussel farm in the Magdalen Island, Quebec in 2004.

polychaete *Capitella capitata* (or similar species in other areas of the world) is tolerant of high organic enrichment, and low oxygen conditions (Tsutsumi, 1990; Pearson and Black 2001). The absence of *Capitella* species in "healthy" or control areas has been ascribed to its poor ability to compete against other infauna in low organic input areas (Pearson and Rosenberg, 1978). The number of scavengers and/or predators may also be greater under shellfish farms. Mussels provide food to small gastropod scavengers (e.g. gastropods in

Grant et al., 1995), and a reef-like habitat for small fish and mobile fauna (Inglis et al., 2000). Furthermore, mussel farms also tend to attract or increase the productivity of large number of predatory fish, starfish, crabs (Mattsson and Lindén, 1983) and winter flounder and starfish (D'Amours and Archambault, 2005). Many of the species observed are of commercial importance. D'Amours and Archambault (2005) observed an increase in the abundance of megafauna at mussel farms compared to transects that lead away from farm sites. The increase in megafaunal density could be the result of an increase in productivity or an attraction to these areas. This hypothesis is presently being tested in an AquaNet project (see Table 1.1 in Section 1.1.5). Finally, the last consistent trend shown in Table 2.5 for shellfish effects was a decline in the abundance of large, deep-burrowing species of molluscs (particularly suspension-feeding bivalves). The loss of bioirrigating species (deep-burrowing infauna) could enhance the anoxic conditions caused by organic enrichment. These changes in species composition could occur before a significant change is measured in sediment chemistry (Edgard et al., 2005). In general, the same consistent trend in the similarity and biotic indices have been observed in the literature reviewed by Inglis (2000).

Less consistent changes seem to occur in cases of low impacts (low organic enrichment) or in diffuse effects (over a whole system such as a bay). Species diversity and total biomass declined and an increase in the dominance of particular species (evenness index) was observed under shellfish farms (Table 2.5) No classical response to organic enrichment has been observed with biomass (Grant et al., 1995). Grant et al. (1995) recorded higher biomass under a mussel farm due to the species *Ilyanassa*, which is a scavenger attracted by decaying animal tissue. However, Callier et al. (2005) observed no differences between 0 m and 300 m but enhanced biomass at intermediate distances (Fig. 2.3) along a transect leading away from a mussel farm in the Magdalen Islands, Quebec. Diversity indices and biomass indicators should be interpreted with caution and studies should combine diversity indices with other indices (e.g. indicator species) to better identify changes.

2.2.3 Sampling design

The general issues with regard to designing appropriate sampling programs to detect environmental changes have been widely discussed (Green, 1979; Underwood, 2000; Gillespie et al., 2002; Underwood and Chapman, 2003). Nonetheless, many of the studies in Table 2.5 (31%) used sampling designs that involved confounding factors. For example, a single farm compared to a single control site can only reveal a difference between those two sites. The difference may be due to the aquaculture activity, but this assumes that the two sites were identical in the magnitude of indices being measured before the aquaculture activity and would have remained identical were it not for the aquaculture activity. Any inference as to why the sites may differ is speculation. What is needed is a sampling design with representative control sites and putatively shellfish impacted sites. It is also possible to use an asymmetrical design with *n* control sites and one shellfish farm site (see Glasby, 1997; Archambault et al., 2001). It is also worthwhile to refer to the report from the “Design Standards for Improving Fish Habitat Management Workshop”, organized by DFO in 2001, that brought together scientists and managers from a variety of disciplines and agencies to develop scientific design standards for assessing the effectiveness of fish habitat mitigation and compensation measures (Gillespie et al., 2002).

The spatial scales at which shellfish impacts may occur are not always predictable and may depend on such variables as current velocity, sediment type, etc. If the shellfish impacts are larger than predicted and affect control sites, then the sampling will be useless to detect impacts. A good sampling design would therefore include many (more than 2) spatial scales or distances from shellfish farms. This problem in sampling design was found in 46% of the studies presented in Table 2.5. Several studies (23%) did however sample around shellfish farms but the furthest distance was only 60 m, which may still be under the influence of the farms. Most of these studies did not find differences, or only minimal changes between sites. The problems described here could have profound implications on management decisions (see Underwood 1993a, b, for a complete discussion).

2.2.4 Thresholds

The greatest difficulty is to set a threshold and sometimes the criterion is a “no net loss” or “no change”. A change can always be detected whatever the disturbance is, and its detection will only depend on the sampling effort (Cohen, 1988). Inversely, it is easy to detect no change, deliberately, by using a sampling design with low statistical power (Keough and Mapstone, 1997). To set an adequate threshold, scientists, managers and all stakeholders must together identify the value of acceptable change from reference conditions. The response of indices to the presence of a shellfish farm is likely to be a continuous function of the magnitude of the farming activities. Values of an indicator measured in the field will obviously be a function of farming activities (Fig. 2.5a); see Mapstone (1995) for a complete discussion. Whether we should consider a threshold A or B, the limit where important changes occur depends again on the question. For example, if the question is “no changes in the density of endangered species”, then the threshold needs to be set below level A (Fig. 2.5). If the question is “no loss in the density of a commercial crab species”, the level could be set over that of B. Research in the sector of public health is much further ahead of ecology in terms of choosing thresholds over linear and step functions; see the review by Calabrese and Baldwin (1999). They concluded that if a disturbance-response (dose-response) relationship is continuous, then no natural threshold exists. A value where change is considered to occur must therefore be chosen on this curve. There is also a concave downward curve relationship that has been observed in some studies (see example of Callier et al., 2005). Callier et al. (2005) found that total biomass was low under mussel leases but increased at a certain distance from the site and decreased again to the level of the mussel sites at a reference site (Fig 2.3). Whether or not the increase in total biomass should be considered beneficial is subjective. This type of relationship increases the complexity of fixing a threshold, especially if the sampling design involves only two sampling sites (i.e. under the shellfish farm and at a control site).

The use of thresholds is often based on mean values but it has been shown in many studies that the ecosystem’s response to a disturbance is an increase in variability (Warwick and Clarke, 1990; Caswell and Cohen, 1991; Warwick and Clarke 1993;

Fraschetti et al., 2001; Duplisea et al., 2002; Hartog, 2003; Callier et al., 2005) (see Fig. 2.5b). Warwick and Clarke (1993) found increased variability in faunal assemblages with an increase in organic matter. It is possible to observe no change in the mean values of the indices, although the variability may increase through time, making it impossible to adequately select a threshold (Fig. 2.5b); see Underwood (1991) for a complete review on this problem.

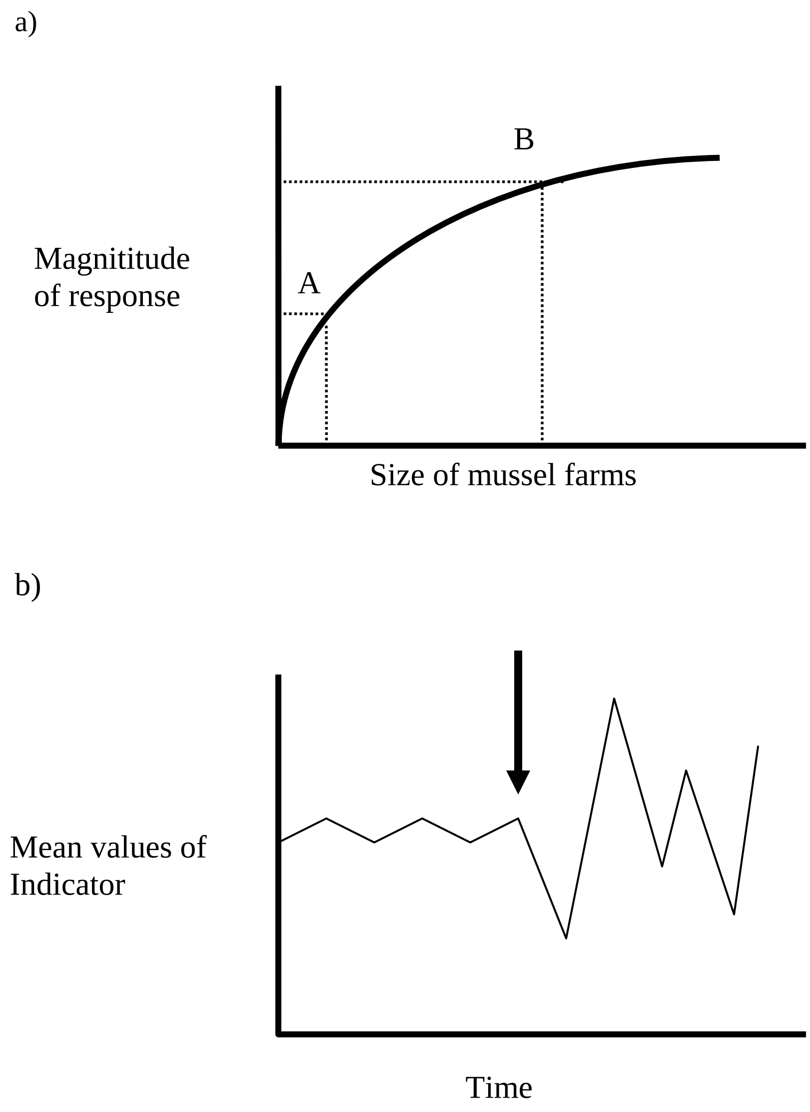


Figure 2.5. a) Hypothetical relationship between the response of indices and the size (strength of disturbance) of mussel farms. The letters A and B indicate two threshold levels (modified from Mapstone, 1995). b) Hypothetical response of the mean value of indices to the beginning of the activities of shellfish farming. The arrow indicates the start of the aquaculture activities.

Scientists can describe the patterns of change in the indices that are selected but what constitutes an important change (negative or positive related to the productivity) is a societal choice involving many parties (managers, the public, cientists, etc.). McDonald and Erickson (1994) propose “bioequivalence”, whereby a threshold is chosen based on a conservative “default” percentile in the range of values exhibited by the selected indicator, and then revising that value based on local knowledge and experience. In this kind of process, all stakeholders should negotiate what constitutes an acceptable threshold or level of change that needs to be detected before undertaking the monitoring process or implementing the project. But as mentioned by Archambault et al. (2001), the use of bioequivalence is not common in biological science and deserves more attention.

2.2.5 Recommendations

1) Indices should be efficacious and relevant to the question tested. They should be associated with the potential impact of shellfish, ecologically and socially important and efficient to measure. Among the indices described in Table 2.5, it is suggested that the *structure of the benthic community* could be monitored based on the *density of opportunistic and/or macrofaunal organisms*. The choice of appropriate indices will depend on the question. A design can be made more economical by reducing the taxonomic resolution with which biota are recorded, thereby reducing the time and cost of sorting multi-species samples. This has been done in many studies (Somerfield and Clarke, 1995; Lasiak, 2003; Cusson et al., accepted). Furthermore, if the large macrofaunal organisms are selected, it could simplify the identification process. These large species are easy to sample with video, by SCUBA divers, small beam trawl (1 m wide) or fixed cages. Usually the data are available in less than a week following the sampling.

2) There are two components to setting a threshold. The first is to characterize how the selected indices are likely to respond to change in the case of the different sizes and types of shellfish farming. The second involves incorporating societal values into deciding how important a change should be before it is unacceptable. Furthermore, a threshold should be based on a solid scientific background and should use an approach similar to the “bioequivalence” of McDonald and Erickson (1994).

3) Optimizing and performing the sampling based on an adequate design is a very important component of any decision-making process. An inadequate sampling design could result in misleading results. An appropriate sampling design will have more than one control site located at various distances from the shellfish farm. It is important to remember that a sampling design with only one treatment site and one control site precludes the conclusion that a measured effect is caused by the treatment because two separate populations may diverge or converge through time without any anthropogenic impacts. Sufficient replication through time is also necessary to detect temporal changes in variance of indices. A change in indicator variance is a better indicator of impact than a change in the mean values since a population may, for example, maintain the same mean abundance of organisms but the variance may increase over time. This increase in variance could eventually cause extinction.

2.3 Bottom Video Indicators and Thresholds (G. Bugden)

2.3.1 Introduction

One of the primary ways that suspended shellfish aquaculture may modify the ecosystem is by increasing the downward flux of organic matter. By filtering suspended organic matter and changing the packaging to larger, more rapidly sinking particles (feces and pseudo-feces), the shellfish can enhance the flux of organic material to the bottom. In depositional environments, this increased flux can result in a significant organic enrichment of the sediments beneath the culture operation, increasing sediment oxygen demand and, in extreme situations, enhancing the risk of bottom anoxia.

Selected chemical measurements in surface sediments, e.g. total sulphide (S) and redox potential (Eh), have been found to be quite sensitive to organic enrichment (Hargrave et al., 1997; Section 2.1). The determination of these parameters entails chemical analysis and the use of delicate probes that require frequent calibration. There may be considerable variability in Eh and S values over small spatial scales, particularly in oxic sediment, and the measurements cannot be made on all bottom substrates. A cobble bottom, for example, is not suitable for Eh/S measurements.

One visual method for detecting the hypoxic-anoxic transition in marine sediments is the presence/absence of white sulphur bacteria mats. *Beggiatoa* spp. are chemoautotrophic anaerobic bacteria that obtain energy by oxidizing H_2S and at the same time reducing NO_3^-. These micro-organisms grow at the redox potential discontinuity (RPD) where Eh changes from positive to negative potentials. When the RPD occurs at or close to the sediment-water interface, chemoautotrophic bacteria, such as *Beggiatoa*, form a white mat over the sediment surface. In shallow water where light reaches the bottom and negative Eh potentials occur at the sediment surface, photosynthetic bacteria such as purple sulphur bacteria may also be present. Therefore, white and purple sulphur bacterial mats on the sediment surface are a clear indication that hypoxic-anoxic sediments are present. The presence of different amounts of S in sediments can also be detected by colour changes. Oxic sediments are usually light brown (tan) or grey since S-metal complexes are not formed. Hypoxic deposits are medium grey to dark brown. Highly reduced, anoxic sediments are dark grey to black due to FeS formation. These visual indications of sediment hypoxia-anoxia suggest that the analysis of colour imagery of the sea floor beneath shellfish culture operations might provide a method for determining the degree of organic enrichment associated with shellfish culture.

2.3.2 Image collection and analysis

During two surveys, one in August 2002 and another in August 2003, colour video images of the sea floor were taken at various locations in Tracadie Bay, PEI, an area of extensive mussel culture. The still images used in the analysis were captured from an inexpensive underwater video system. The video camera was mounted in a frame that permitted determination of the distance from the camera to the bottom, allowing standardization of the area encompassed by the image (Fig. 2.6). Images from both 2002 and 2003 were grouped for image analysis under the assumption that the bottom characteristics would not have changed much over this interval. The objective of this analysis was to improve upon more subjective previous efforts at bottom characterization using underwater video by developing a more quantitative analysis approach.

Figure 2.6. Underwater video camera showing the 0.5m x 0.5m image frame.

Video footage was recorded in the field on magnetic tape using a digital video camera in VCR mode. The position, date and time from a differential GPS receiver were imprinted on the video using a GPS screen overlay. Later, in the laboratory, representative still images for each station were captured from the tape using video processing software on a personal computer. The still image frame was generally chosen as the moment when the camera frame contacted the bottom but before the disturbed sediment had a chance to cloud the picture. This insured that the area covered by the image was quantifiable. The selected images were then imported into image processing software designed for remote sensing applications. The images were cropped to remove the overlain navigation information and the frame, leaving an image representing an area of approximately 30 cm x 40 cm for analysis. The images were then enhanced using a linear enhancement program and subjected to a supervised maximum likelihood classification with a null class, which resulted in the assignment of each image pixel to one of four groups on the basis of its Red/Green/Blue values (Fig. 2.7). A supervised classification requires the operator to choose training areas representative of each of the identified classes within the image under analysis. In general, three training areas were chosen to represent: 1) *Beggiatoa* mats; 2) reduced sediment; and 3) un-reduced sediment. Image pixels that were not assignable to any of the three training areas were designated as “null”. The null pixels generally represented objects such as mussel shells, twigs or eel grass strands, and were not included in the calculations of the fractional areas. A small portion of the images did not contain all three classes. This situation was dealt with by using a smaller number of classes and assigning the missing class(es) a count of zero pixels.

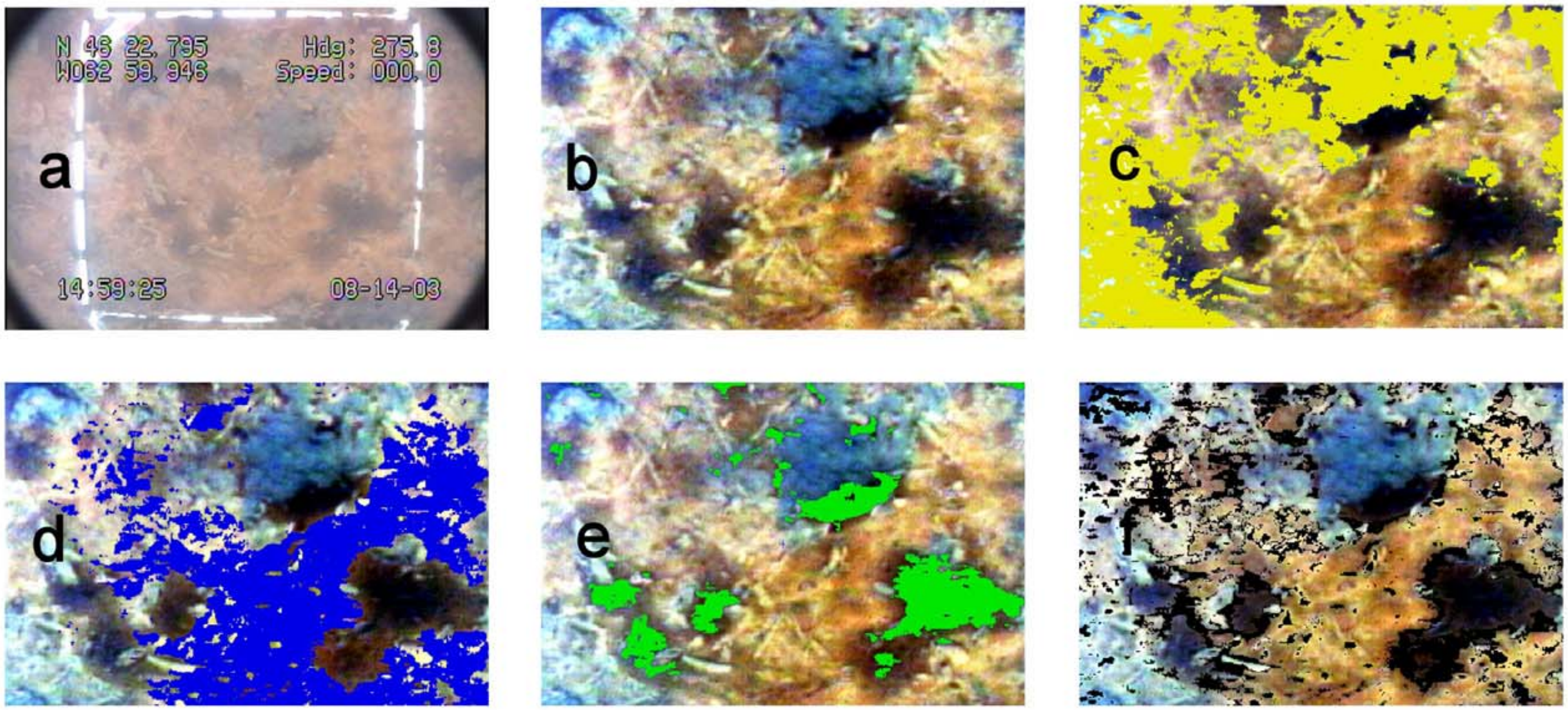


Figure 2.7. Sample image classification: a) captured image; b) cropped and enhanced; c) *Beggiatoa* mat class (Yellow: 36.8%); d) un-reduced sediment (Blue: 33.4%); e) reduced sediment (Green: 9.6%); f) null class (Black: 20.2%). All percentages are given as fractions of the whole image.

2.3.3 Results of analysis

To estimate the stability (i.e. precision) of the analysis, a series of 5 non-overlapping images of the bottom located within a radius of 25 m of each other were processed and the results compared. If the image frame was smaller than the scale of variability of the three classes identified, the results of the analysis would vary wildly, indicating that a single image was not representative of bottom characteristics on the scale of a typical aquaculture lease. The results are shown in Table 2.6. The analysis results are seen to be reasonably stable, indicating that the approximately 30 cm x 40 cm area of the image is of sufficient size to adequately represent sediment characteristics on the scale of a lease in the survey area.

Table 2.6. Results of the analysis of 5 non-overlapping images collected within a 25 m radius of each other. Analysis indicates that estimates of the area covered by each bottom class are precise and are therefore representative of bottom characteristics on lease scales.

	Fraction of classified pixels			**Fraction of image**
	Reduced	**Unreduced**	***Beggiatoa***	**Null**
	0.28	0.28	0.45	0.07
	0.37	0.25	0.39	0.07
	0.29	0.20	0.52	0.20
	0.53	0.27	0.20	0.15
	0.35	0.23	0.42	0.14
Average	0.36	0.24	0.39	0.12
Std. Dev.	0.10	0.03	0.12	0.05
N	5	5	5	5
Std. Err.	0.05	0.01	0.05	0.02

Lack of water clarity during the acquisition of the imagery was a significant problem. Of the approximately 32 images that were felt to be separated by enough horizontal distance to be representative of different benthic conditions, only 18 were considered to be of suitable quality for further analysis (Fig. 2.8). The results of the image analysis, along with some basic statistics, are presented in Table 2.7. Perhaps the most interesting result from the analysis is that *Beggiatoa* mats are present in all but two of the images, one of which is from a tidal channel with a hard sand bottom. This would indicate that hypoxic-anoxic conditions in the near-surface sediments are widely distributed in Tracadie Bay, as might be expected given the the relatively limited tidal exchange and the organic loading from natural sources such as eelgrass beds, upstream agriculture, and shellfish aquaculture. A bottom video survey of eight PEI mussel aquaculture embayments was conducted in August, 2001, and included many of the sites (mussel lease and reference) previously sampled by Shaw (1998). *Beggiatoa* mats were observed at 11 of the 20 lease sites surveyed, and at 2 of 19 reference sites (P. Cranford, unpublished data).

Table 2.7 Correlation matrix: **Begg** = Fraction of image classified as *Beggiatoa* spp.*;* **Red** = Fraction of image classified as reduced sediment; **!Red** = Fraction of image classified as non-reduced sediment; **Den** = Density of culture (metric tons per hectare); **U** = RMS tidal current from numerical circulation model (cm/s); **T** = Period lease under cultivation (years); **Eh** = Redox potential; **S** = Total sulphides; and **N** = Number of samples.

	Begg	Red	!Red	Den	U	T	Eh	S
Begg	1.00	**-0.41**	**-0.77**	0.05	0.14	**0.35**	0.21	-0.14
Red		1.00	-0.27	**0.45**	-0.03	0.15	0.33	0.02
!Red			1.00	**-0.37**	-0.12	**-0.48**	-0.39	0.12
Den				1.00	0.10	-0.15	0.21	-0.07
U					1.00	**-0.34**	-0.09	**0.61**
T						1.00	0.30	-0.30
Eh							1.00	**-0.79**
S								1.00
N	14	14	14	14	14	14	10	10

Figure 2.9 shows output from a numerical circulation model of Tracadie Bay. Shown is the Root Mean Square (RMS) current speed during spring tide conditions, a measure of the magnitude of the tidal currents. The size of the tidal currents might be expected to be inversely related to the degree of hypoxia-anoxia in the surface sediments. Stronger currents could possibly disperse feces and pseudo-feces over a wider area, reducing localized organic enrichment from shellfish culture, ventilate the bottom with oxic water and, if large enough, physically disrupt the *Beggiatoa* mats, which are quite fragile. Comparison of Figs. 2.8 and 2.9 does not immediately indicate that the RMS current is a major factor in determining the distribution of *Beggiatoa* mats in Tracadie Bay.

Other factors, in addition to the currents, that might be expected to be related to the degree of organic enrichment of the sediments under a culture site are: the stocking density; the length of time the site has been used for culture operations; and the depth of the water. As seen in Table 2.7, the water depth at the image analysis sites is relatively uniform and will be excluded from further analysis. The correlation matrix shown in Table 2.8 was developed to explore the relationship of the image analysis results to lease characteristics, current speed and Eh/S measurements. The length of time between the issue of a site lease and mid-August 2003 was derived from a Husbandry Database provided by Luc A. Comeau (personal communication). Stocking densities were derived from the same database and were representative of the situation a few years prior to the acquisition of the video images. The Eh/S measurements were provided by Barry Hargrave (personal communication) and were from sediment samples collected primarily in 2002. The circulation model current speed was taken from the model grid point nearest

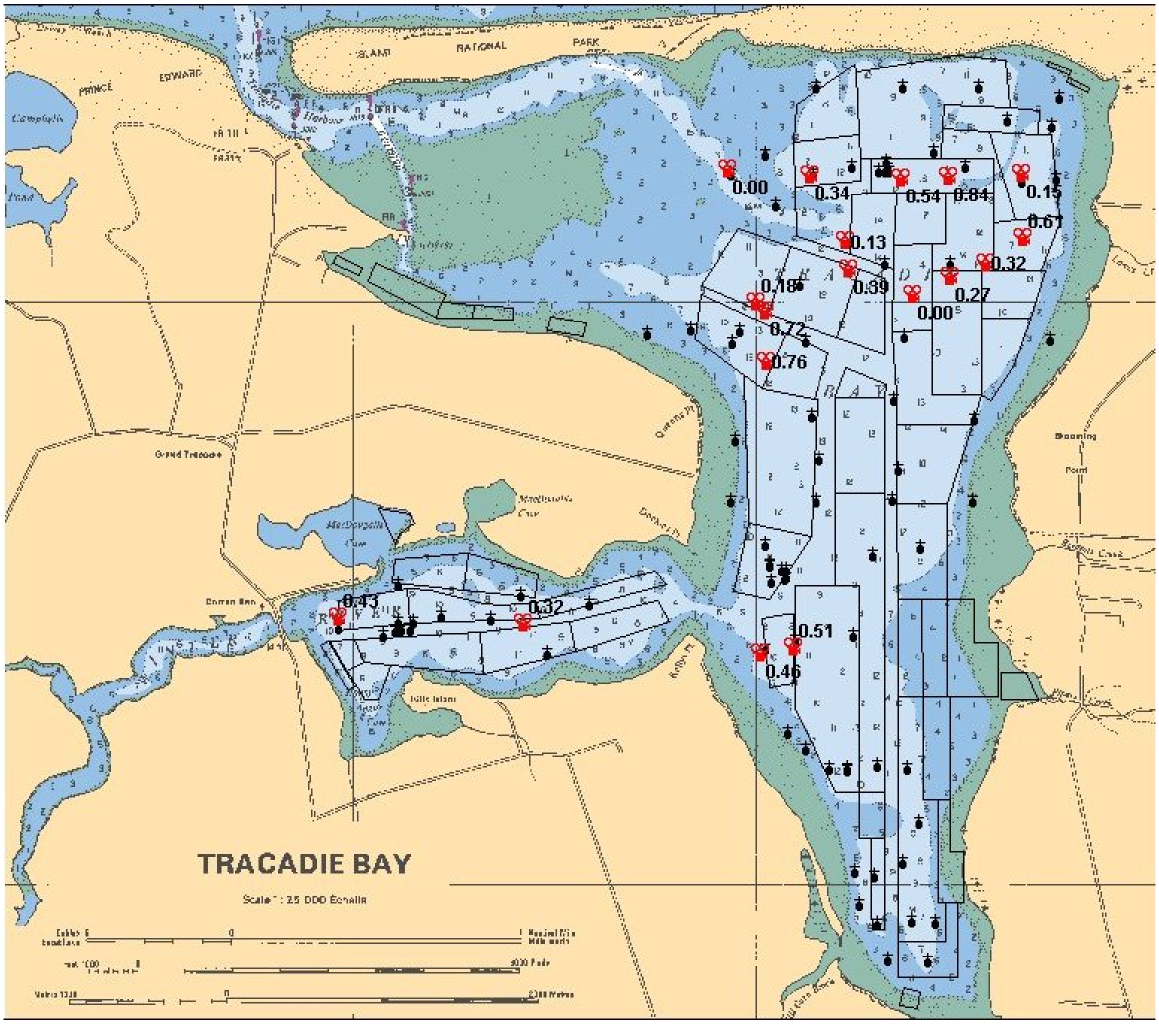


Figure 2.8. Chart of Tracadie Bay, PEI, showing lease boundaries, camera stations selected for image analysis (annotated with fraction of image area covered by *Beggiatoa* mats) and stations where sediment samples were collected for Eh and sulfide analysis.

the video station. The Husbandry Database parameters were taken from the lease which contained the video station. The Eh/S measurements were taken from the nearest sediment sampling station within the same lease. As this was an *ad hoc* comparison, not all leases were represented in the Husbandry Database and not all video stations had a sediment sampling station close enough to be representative. This resulted in a further reduction in sample size to 14 images for comparison to the husbandry variables and 10 for comparison to the sediment parameters. This small sample size results in poor statistical confidence in the correlation coefficients. The only correlations that are statistically different from zero at the 95% confidence level are those between *Beggiatoa* coverage and non-reduced sediment coverage, and between Eh and total sulfides. Both relationships are expected as artefacts of the analysis method or well-understood chemical relationships. However, by looking at the statistics from a different perspective, some tentative conclusion can be drawn. Using standard statistical calculations, it can be shown that it is 80% certain that the correlation coefficients shown in bold type in Table 2.8 have the correct sign (Hopkins, 2000). That is, the odds are that approximately 4:1 that the indicated coefficient is greater than 0.1 if positive, or less than -0.1 if negative.

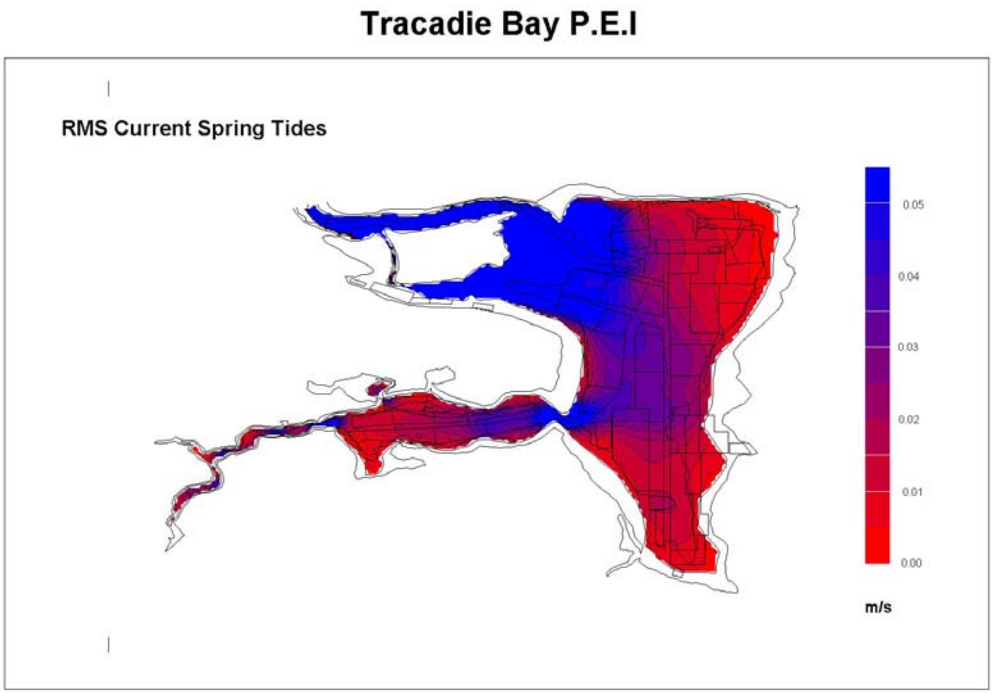


Figure 2.9. Output from a numerical circulation model of Tracadie Bay PEI showing the RMS tidal current.

As discussed briefly above, the modeled tidal current speeds do not appear to be related to any of the parameters derived from the image analysis. The current speed appears to be negatively related to the time the lease has been under cultivation and positively related to the total sulphide concentration. The high positive correlation with the sulphide concentration is difficult to explain and may be spurious. The smaller negative correlation with the duration of cultivation might be due to the more recent expansion of culture into the less sheltered areas and areas closer to the tidal channels at the north end of the bay and the entrance to Winter Bay where the currents would be stronger. Apart from the difficult to explain positive correlation between the tidal current and the sulphide, this analysis did not suggest any relationship between either Eh or sulfide and any of the video-derived sediment. The negative relationship between Eh and sulphide is as expected based on well-understood chemical principles (see Section 2.1).

Encouragingly for the purposes of this study, the data suggest that the variables derived from the image analysis are related to those derived from the husbandry database in a manner that would be expected if the shellfish culture were resulting in an increased flux of organic matter to the bottom. The fraction of the bottom covered by *Beggiatoa* mats appears to be positively related to the length of time the culture operation has been operating. The fraction covered by reduced sediment seems to be positively related to the density of the culture and the fraction consisting of un-reduced sediment is negatively related with both stocking density and period of operation.

Figure 2.10. Sample video frame. This image is anomalous in the sense that there are no *Beggiatoa* mats and many empty shells indicative of drop-off. The vast majority of the other images showed extensive bacterial mats and no shells were visible.

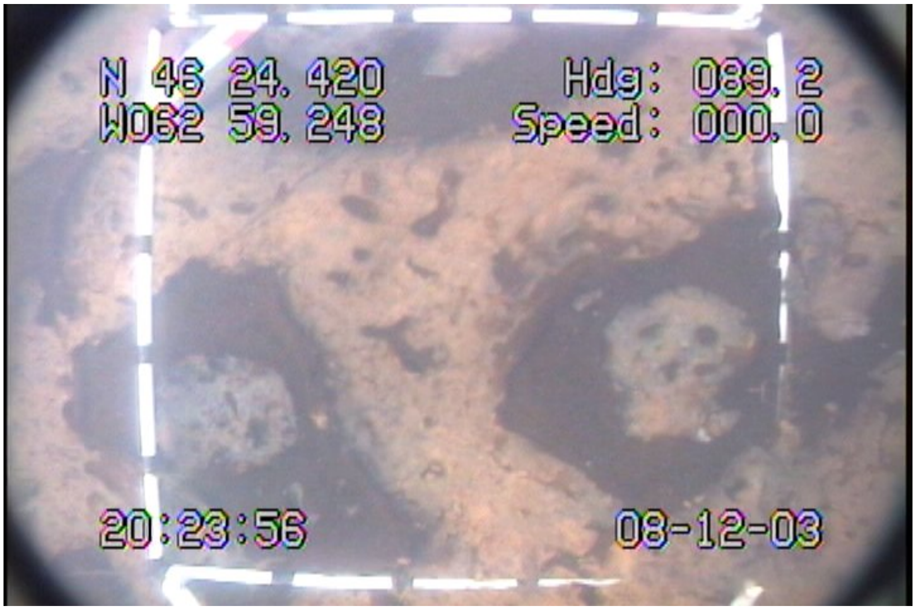


Figure 2.11. Sample video frame. Large portions of the bottom of Tracadie Bay appear very similar to this image. Note the features with the central *Beggiatoa* mat and surrounding annulus of reduced sediment; these have a diameter of about 30 cm.

2.3.4 Summary and recommendations

Bacterial mats indicative of hypoxic-anoxic conditions at the sediment-water interface are present to varying degrees over the entire area of Tracadie Bay deeper than about 4 m. It is uncertain to what extent this is a natural phenomenon or to what extent the situation may have been exacerbated by agricultural inputs and shellfish aquaculture, as observations of the undisturbed ecosystem are not available (but see Working Paper #4). Unexpectedly, the distribution of the tidal currents does not seem to play a major role in the determination of the extent of *Beggiatoa* mats or reduced sediment. It may be that, away from the tidal channels, the near-bottom currents are simply too small to make a significant difference. There are some indications, from the correlation analysis, that shellfish aquaculture has had an impact on the characteristics of the sediment in Tracadie Bay in a manner consistent with an increase in the flux of organic matter to the bottom. Both stocking density and length of time under cultivation seem to be appropriately related to parameters such as the fraction of the bottom covered by sulphur bacteria mats and the fraction covered by reduced and un-reduced sediments.

At least in the study area, the approximately 0.5 m x 0.5 m image frame size was sufficiently large to include variations in bottom features so that a single image was indicative of conditions on the scale of a lease. Sediment characteristics varied from bacterial mats, to reduced sediment, to un-reduced sediment, on a scale of a few tens of cm, probably due to subtle variations in bottom topography (Fig. 2.11). This horizontal scale is of the same order of magnitude as the size of the devices often used to sample sediment for subsequent Eh/S analysis and may explain some of the scatter in these observations (see Section 2.1). Water clarity can be a significant problem but can be overcome by judicious choice of sampling time.

The use of benthic imagery to characterize changes in the bottom beneath shellfish culture operations would appear to be a useful technique. The method may be applicable to all substrates and degrees of impact. The present analysis focused on symptoms of surface sediment anoxia, as those were the conditions encountered in the study area. It is possible that in other environments the enumeration of features such as worm holes might be a more appropriate measure. The indicator and threshold will vary with substrate, depth and other conditions. The analysis of the imagery should be as quantitative as possible and this, at the very least, requires knowledge of the area encompassed by the image under analysis. Further site-specific analysis should be conducted to determine if a single image is resolving variations on the scale of a lease or if multiple images are required. The image analysis may be mathematically sophisticated, as in the present study, or simply consist of overlaying a transparent grid and making visual counts. Regardless, a sequence of images acquired before culture is initiated would be very useful for separating the role of the shellfish culture from other natural and anthropogenic processes.

2.4 Hydroacoustic habitat classification indicators (M. Ouellette)

The objective of this section is to discuss the potential applicability of acoustic habitat classification systems and specific acoustic habitat indicators for assessing benthic impacts from shellfish aquaculture. This discussion is based on results obtained from an acoustic survey, using a single beam seabed classification system, of a mussel farming bay (see Case Study 3 in the Habitat Sensitivity paper for more details). Reviews of the various acoustic tools and techniques available for the acquisition of benthic habitat mapping data are available in several publications (Kenny et al., 2003; Waddington and Hart, 2003; Diaz et al., 2004).

2.4.1 Introduction

A basic knowledge of seafloor habitats is necessary for the development and implementation of a wide variety of resource management policies. The need to efficiently assess and monitor benthic habitats in the nearshore and estuarine zones is becoming increasingly evident to the various agencies that are involved in coastal zone management.

Benthic habitats can be defined as submerged bottom environments with distinct physical, geochemical and biological characteristics. These habitats vary widely depending upon their location and depth, and they are often characterized by dominant structural features and biological communities (Diaz et al., 2004). Estuarine and nearshore benthic habitats can be highly diverse, including shallow submerged mudflats, rippled sandflats, rocky hard-bottom habitats, seagrass beds and shellfish beds. The mapping of benthic assemblages has proven to be challenging on a large scale.

Benthic habitat mapping is a multidisciplinary task that combines physical (geological), biological, oceanographic and chemical components of the seafloor. Data such as substrate type, topography, biological species and oxygen concentration are all necessary to create an accurate picture of a habitat (Diaz et al., 2004). The acquisition of benthic habitat data is typically a costly and time-consuming effort. However, advances over the last decade in technologies and disciplines associated with the field of geomatics are showing high potential for the acquisition and analysis of some of the data layers needed in benthic habitat mapping. Geophysical techniques that help identify and define large-scale marine benthic features are valuable in appraising essential habitats of marine benthic assemblages. Most importantly, these technologies are capable of providing accurate and repeatable measurements. These are critical requirements for measuring spatial and temporal variations of the seabed that could be associated with anthropogenic activities.

The acoustic method (single beam sonar) of remote sensing data collection is an accurate, low-cost, and relatively simple technique for generating seafloor topography and for characterizing the surface sediment composition (with acoustic seabed classification systems), especially in areas with gradual seafloor relief or shallow water depths. In all acoustic systems, an increase in frequency leads to an increase in resolution and a decrease in range or depth of coverage (Waddington and Hart, 2003). Given their various configurations, acoustic systems should be selected in accordance with the specific objectives of the benthic habitat mapping program (e.g. higher frequencies for submerged aquatic vegetation [SAV] mapping, medium frequencies for epibenthic mapping and lower frequencies for endobenthic mapping).

One advantage of single beam echosounders is the ability to interface them with seabed classification coprocessors. Acoustic seabed classification is the organization of seabeds into discrete units based on a characteristic acoustic response. The echo waveform shape is a measure of the acoustic energy (or backscatter) redirected to the echo sounder transducer. The signal amplitude and shape is influenced by physical attributes of the surface sediments and immediate subsurface. The seabed characteristics that have a major influence on the signal include: sedimentary properties of the substrate that can affect hardness (echo penetration); seabed roughness (echo scattering); and biotic communities living on or in the seabed (Preston and Collins, 2000). The limitations of single beam echosounders are generally associated with the narrow swath width of the transducers that makes it difficult to conduct a continuous coverage of the seafloor. The output resolution of the acoustic data is determined by the footprint size of the echo (which varies with depth), the sampling interval along the track lines (influenced by the sampling speed of the system and the speed of the survey vessel), and the distance between transects (von Szalay and McConnaughey, 2002). However, a large acoustic footprint could result in a greater averaging of seabed features and reduced ability to resolve boundaries in acoustic seabed classification (Collins and Rhynas, 1998). All these factors are important in determining the accuracy of the final map, i.e. by considering the amount of spatial interpolation needed between data points to generate a full-coverage of a given area.

Any comprehensive seafloor characterization effort will generally rely on some combination of broad-scale, lower resolution, physical characterization data (e.g. multibeam bathymetry, side-scan sonar imagery, etc.) as well as fine-scale, higher resolution sampling data (e.g. sediment grabs, sediment-profile imaging and underwater video). The broad-scale techniques are intended to provide a general physical overview (e.g. bottom topography and changes in surface sediments) of the seafloor over the entire area of interest. The fine-scale techniques are used to generate the higher resolution, ground-truth data that will improve and/or confirm the broad-scale interpretation (Waddington and Hart, 2003).

The key to successful application of this technology, however, lies in the translation of basic physical data on bottom substrate and characteristics into meaningful representations of benthic habitat quality (Diaz et al., 2004). The physical characterization of the seafloor is undoubtedly one of the most important elements in any comprehensive benthic habitat classification scheme (Waddington and Hart, 2003).

2.4.2 Indicators and thresholds

An example of an acoustic seabed classification map of a mussel culture bay is shown in Figure 2.6 for Tracadie Bay, PEI, a shallow (mean of 3 m with a maximum of 6 m water depth), nearly enclosed tidal lagoon (surface area of 14 km^2) located on the north shore of PEI. The general oceanographic characteristics of Tracadie Bay are shared by a number of bays on the north shore of PEI and the Gulf coast of New Brunswick (Dowd et al., 2001). The acoustic data to generate this map were obtained during surveys conducted in 2002 and 2003, using a single beam QTC View-V shallow water sonar system (50 kHz frequency with a beam width of 24°) (M. Ouellette, unpublished data). The approach for applying this system to seabed classification involves three steps: (1) echo digitization of the first returning echo during data acquisition; (2) echo description by the application of algorithms to analyse and generate a series of features; and (3) echo classification where the most useful features are chosen by principal components analysis and assigned an acoustically distinct class representing the seabed (QTC Impact, 2004).

The QTC View-V type of system was shown to be efficient for mapping bathymetry and certain substrate features in Tracadie Bay. Post-processing of the data, using the waveforms editor in the QTC Impact (3.4) software, allowed us to eliminate water column interferences (such as ropes, mussel socks and buoys) from the dataset before conducting the seabed classification analysis (QTC Impact, 2004). Furthermore, the results were consistent between the two surveyed years, which demonstrates the capacity of this system to provide accurate and repeatable measurements. This approach is therefore well suited for the measurement of spatial and temporal variations (monitoring) in some physical characteristics of the seafloor.

Preliminary results (Fig. 2.12) for the habitat characterization efforts of the various seabed classes show that one class (red) is generally associated with flat bottoms of very soft mud, with very little or no SAV and very few epibenthic fauna. A second class (yellow) is also associated with relatively flat bottoms, but with more consolidated mud, the presence of some SAV (not always) and more benthic fauna (including the presence of bacterial mats in some areas). The remaining classes (green and blue) are associated with more complex benthic habitats (bottom not always flat, sand-mud substrates, the presence of denser SAV [not always], and the presence of endobenthic and epibenthic fauna). The first and second classes are generally associated with the deeper part of the bay, whereas the others are associated with the shallower and more dynamic shoreline (Dowd et al., 2001).

There is no obvious relationship between the distribution of the acoustic habitat classes and the location of mussel culture leases in this bay. The depth (volume) of the acoustic measurement in the substrate at low sonar frequencies, such as 50 kHz, is mainly subsurface (several centimetres), depending on the type and state of the sediments (Collins and Rhynas, 1998; Preston and Collins, 2000). This suggests that the physical changes in the substrate that could be associated with mussel culture leases would be at a more superficial layer. This will be investigated, now that the current system is also equipped with a higher frequency band (200 kHz) which, in theory, should be capable of generating an acoustic map that shows more of the superficial layer of the seabed.

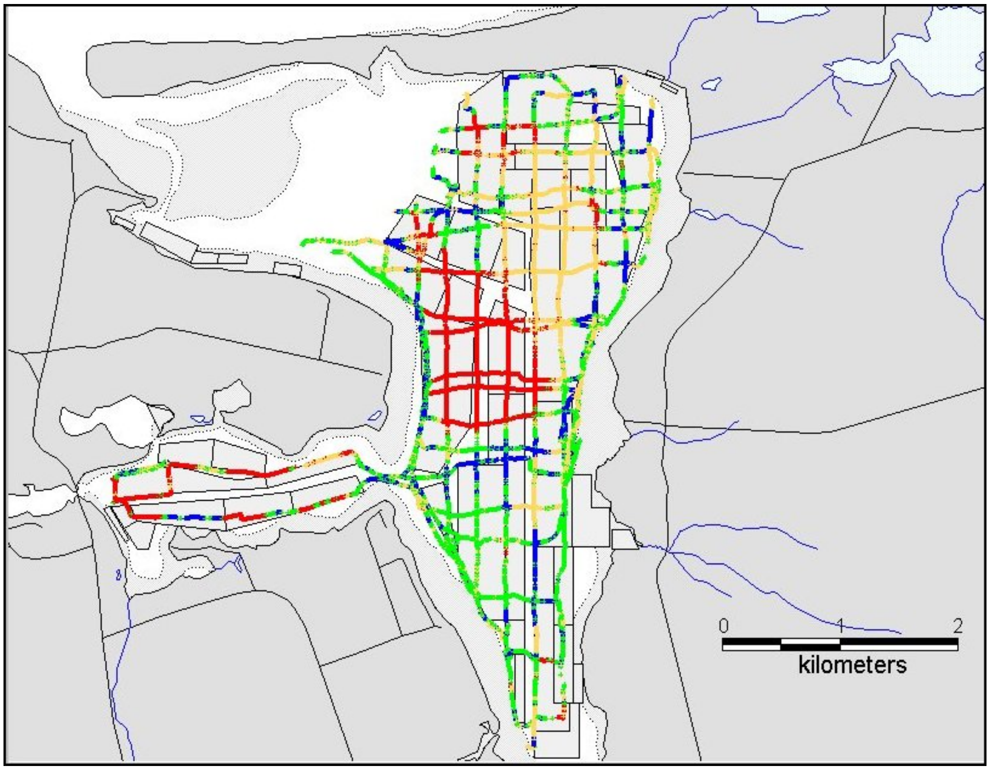


Figure 2.12. Acoustic seabed classification map of Tracadie Bay, PEI. The acoustic data were obtained during a survey conducted in 2002 and 2003, using the QTC View-V (50 kHz frequency with a beam width of 24°). Mussel culture leases are represented by polygons underneath the survey transect lines (M. Ouellette, unpublished data).

The pertinent question from Habitat Management in this process would be: "can we use this technique to measure an indicator of the interactions of mussel culture activities with the benthic habitat?", and if yes, "what are the thresholds where significant changes can be identified?" In order to identify an indicator and possible thresholds associated with this acoustic measurement, we would need to know exactly what features of the seabed had the most influence on the acoustic signal and thus classification. This remains a challenge, mostly because the seabed parameters that can influence the acoustic signal are numerous, complex and variable. For example, where infaunal invertebrate species have particularly strong impacts on sediment structure (bioturbation), acoustic methods could prove useful in locating nursery grounds and habitats containing large species, but provide little assistance in understanding fine-scale species interactions or identifying the factors controlling assemblage structure (Solan et al., 2003).

The type of substrate (sediment grain size) and the state of that substrate (e.g. shear strength and porosity) have a major effect on echo penetration. The substrate topography (such as flat bottom, sand ripples, rocks and slopes), and the reflectivity of that material, will have an effect on echo scattering (Preston et al., 1999; von Szalay and

McConnaughey, 2002; Kenny et al., 2003). The biotic community (endobenthic and epibenthic) and patchiness can also have a major influence on the acoustic signal. In theory, the structures (e.g. shells) of benthic fauna could influence the acoustic reading if the size and/or density of individuals are great enough. The burrowing activities of several species could also have a significant influence on the state of the substrate (Gray, 1981). The type and densities of SAV (e.g. eelgrass beds), caused by the acoustical reflectivity of the gas-filled plant stems and/or blades and biogenic accumulations (e.g. bivalve reefs), can also influence echo scattering (Sabol and Johnston, 2001). This complex assemblage of physical and biotic communities, along with the chemical particularities, is what constitutes the benthic habitat that we are trying to map.

2.4.3 Summary and conclusions

Acoustic remote sensing systems can efficiently be used to collect bathymetry data that can be interpolated to generate a continuous topographic map of the seafloor. This layer of information is crucial in any comprehensive benthic habitat mapping project. Acoustic seabed classification systems presently available on the international market from several companies vary in their approach to the acquisition and analysis of single beam acoustic data. For example, some systems analyse the first return echo while others analyse the first and second returns of the sound wave. Data sharing is thus presently hampered by the lack of uniformity and standards in data collection, classification and processing protocols (Preston and Collins, 2000; Kenny et al., 2003; Diaz et al., 2004). A single beam sonar system has proven to be useful in mapping relatively shallow bays and estuaries, including bays with extensive mussel culture activities. This system, when interfaced with acoustic seabed classifiers, can analyse the returning echo for various features of the seabed, such as substrate hardness and roughness. These data can then be used to characterize the substrate composition of the seabed, another important layer in benthic habitat mapping.

The acoustic seabed classification system used in this survey of an extensively leased shellfish aquaculture inlet in PEI was able to generate accurate and repeatable measurements of the seabed over the two years. This suggests that this tool could be used to assist in monitoring spatial and temporal changes associated with the acoustic physical characteristics of the seabed. However, there was no obvious relationship between the acoustic data obtained in Tracadie Bay and the location of mussel culture leases in this bay. Given that a low sonar frequency was used in this survey, any potential physical changes of the substrate associated with mussel culture leases may have been at a more superficial layer. Additional research is needed to ground truth habitat characterizations based on acoustic surveys before potential indicators and thresholds for use in assessing and/or monitoring marine shellfish culture activities can be identified. Diaz et al. (2004) did an extensive review of the indices of aquatic habitat quality suggested over the last 20 years and found that there is little acceptability of any specific metric by environmental managers or scientists. Owing to the importance of seafloor characterization fornumerous nationally important applications (e.g. nautical charting, navigation safety, dredge monitoring, commercial fishing, coastal engineering and benthic habitat assessment), there is a great deal of government-funded research and development underway to improve various aspects of the acquisition and interpretation of seafloor characterization data (Waddington and Hart, 2003).

In all cases, the link between the acoustic waveform analysis and the seafloor classification scheme must be based upon extensive ground-truth data (e.g. grab samples and video) obtained from various seafloor types likely to be encountered. This established relationship between the acoustic waveform and the seafloor type is very dependent on both the echo sounder settings (e.g. frequency, power and gain) and seafloor type. This relationship needs to be re-established for each new project area, or anytime the echo sounder settings are modified (Waddington and Hart, 2003). The problem of data density mismatch between physical and biological methods will likely not be solved until acoustic methods can routinely resolve the elusive biological components that make a physical substrate a habitat (Diaz et al., 2004). This is required to ensure that the acoustic classes identified are biologically relevant.

3.0 PELAGIC HABITAT

3.1. Food particle depletion (P. Cranford, G. Bugden, E. Horne)

Particle depletion is defined here as a significant reduction in suspended particulate matter resulting from consumption by cultured shellfish. The word “depletion” should not be perceived as having a negative connotation as food consumption by any filter feeder results in some level of depletion. Particle depletion is only of concern when large populations of introduced organisms remove food particles faster than tidal exchange and primary production can replace them, resulting in a significant reduction in the particulate food supply for extended periods and over relatively large (e.g. lease to bay-wide) scales. Alteration of the particulate food supply of this magnitude has important implications for the productivity of cultured shellfish (e.g. negative feedback) as well as that of other resident organisms sharing the same food resources (e.g. zooplankton and wild benthic filter feeders). Food depletion is therefore closely linked to the concepts of production and ecosystem carrying capacity.

The capacity for shellfish to deplete particles is controlled, in part, by the efficiency of the gill to capture particles. Suspension feeding bivalves are able to retain suspended particles larger than 3-7 µm with 100% efficiency (varies with species). There is a steep decline in retention efficiency below this size range and less than 50% of 1 µm particles are retained by mussels and oysters (Møhlenberg and Riisgard, 1978). Most picoplankton (0.2 to 2.0 µm) are therefore not effectively captured as a food source by bivalve filter feeders. The upper limit to particle consumption by shellfish is between 0.5 and 6 mm (Karlsson et al., 2003), which includes mesozooplankton (100 to 1000 µm).

Particle depletion by wild and introduced shellfish populations is believed to be greatest in estuaries and inlets where water residence time is long and shellfish biomass is high (e.g. Dame, 1996). In such areas, water depleted of particles by the cultured shellfish cannot be completely renewed by tidal exchange. Comparisons of water residence times in PEI coastal embayments, with estimates of the time required for mussel cultures to clear this water of particles (clearance time), indicate that food supplies in 12 of the 15 embayments studied are being removed faster than they can be replaced by tidal exchange (Cranford et al., 2003). Meewig et al. (1998) used a mass-balance approach to model phytoplankton biomass in PEI embayments and estimated that the mussel farms in six of 15 systems reduced phytoplankton biomass by 45 to 88%. A model study of Tracadie Bay, one of several extensively leased embayments in PEI, was conducted to understand the control of particle concentrations by biophysical processes (Dowd, 2003; 2005). The results suggest that food supplies are affected by shellfish grazing, but that the magnitude of the effect varies spatially depending on local tidal transport processes.

Anecdotal information from mussel farmers and farm production data (Comeau et al., 2005) indicates a reduction in mussel production and meat yields in Tracadie Bay over the past decade, suggesting that food demand has exceeded the capacity of tidal flushing to replenish the food supply (i.e. production carrying capacity exceeded). Industry observations complement the results of food depletion model predictions. Grant et al. (in preparation) developed an ecosystem model of seston (particle) depletion in dense mussel culture and applied the model to Tracadie Bay. The results indicate that cultured mussels may be responsible for severe bay-wide depletion of phytoplankton biomass which could negatively affect the productivity of all secondary producers, including mussels, through food limitation. Fish habitat alterations leading to a net loss of system productivity are therefore indicated. The development of approaches and tools (indicators) for documenting the degree of particle depletion is critical to maintaining ecosystem productivity, ensuring aquaculture sustainability and managing the coastal zone responsibly.

Several attempts have been made to measure particle depletion at raft and longline shellfish aquaculture sites using a variety of sampling approaches, including water sampling and moored or profiling *in situ* instruments. In general, depletion within the geographic scale of shellfish farms is substantially greater for mussel raft culture (Navarro et al., 1991; Heasman et al., 1998) than for the longline culture method used in Canada (Rosenberg and Loo, 1983; Fréchette et al., 1991; Ogilvie et al., 2000; Pilditch et al., 2001; Ibarra, 2003; but see Strohmeier et al., 2005). Very close spacing of mussel ropes attenuates water flow through the raft system (Boyd and Heasman, 1998), allowing the mussels to clear much of the particle load and thereby depress food supplies and secondary production downstream (Navarro et al., 1991; Heasman et al., 1998). Studies of food depletion associated with longline culture have provided variable results, with no food depletion reported inside some farms (Fréchette et al., 1991; Pilditch et al., 2001), and significant depletions observed inside others (Rosenberg and Loo, 1983; Ogilvie et al., 2000; Ibarra, 2003; Strohmeier et al., 2005). Such variability is expected given site differences in culture density, circulation patterns, current speed and mixing processes. Also, the ability to detect food depletion may be beyond the capacity of some of the methodologies employed.

3.1.1. Indicators and approaches for detecting particle depletion

Potential indicators of particle depletion include turbidity (total suspended particulate matter; TPM), chlorophyll *a* (related to phytoplankton biomass) and measures of underwater light properties such as light penetration depth (e.g. Secchi depth) and light attenuation. These are direct and indirect (underwater light environment) measures of the ambient concentration of suspended particles. The use of any of these indicators for measuring food depletion on scales relevant to aquaculture leases and coastal ecosystems is problematic, as the high degree of short-term spatial and temporal variability in particle concentrations in dynamic coastal systems has a tendency to mask any depletion (Bacher et al., 2003). Sampling approaches based on the manual collection of water are of limited use for quantifying depletion when the water body changes faster (e.g. tidal cycle variations) than the time required to complete the full sampling survey. The selection of a reference (unaffected) site is also problematic since the water is in near-constant motion and is rapidly transported throughout the embayment. As a result, the potential efficacy of manual sampling is limited to studying depletion on relatively small spatial scales.

A variety of moored instruments (e.g. fluorometers, transmissometers and optical backscatter sensors) are available for monitoring all the identified indicators of particle depletion and have greatly increased the frequency at which measurements can be made. However, the requirement to quantify spatial variability (horizontal and vertical) can only be addressed with the use of a large number of these instruments. Ibarra (2003) used moored instruments that measure the difference in the diffuse attenuation coefficient of light between the upper and lower depth limits of a shellfish operation. These measurements provide a depth-averaged indication of particle loads. Several moored instruments are used to quantify horizontal variations within a lease and at a suitable reference site. Since no suitable reference sites are available under conditions where impact assessments predict a bay-wide influence of aquaculture on food particles, the moored instrument approach is limited to lease-scale monitoring of particle depletion.

The detection of multi-lease and bay-scale particle depletion requires technologies that enable the rapid mapping of the water column in two (latitude and longitude) or three dimensions (x, y and depth). A rapid sampling speed is critical to avoid confusing shellfish effects with natural temporal variations related to a moving water body. Two such approaches tested during studies of environmental interactions of mussel culture in Tracadie Bay are: (1) low-altitude remote sensing using the CASI (Compact Airborne Spectrographic Imager) hyperspectral scanner; and (2) high-frequency measurements with a towed, undulating vehicle (BIO-Acrobat) equipped with particle sensors (chlorophyll fluorometer and light attenuation meter). A major strength of the remote sensing approach is the ability to quickly map depth-integrated food supply indicators (chlorophyll) over whole bays, excluding shallow regions where light penetration reaches the seabed. The towed vehicle approach provides a direct measure of particle concentrations and can rapidly survey their horizontal and vertical distribution over large areas. However, it cannot provide the 1-m^2 surface resolution that can be achieved by low-altitude remote sensing.

CASI images of chlorophyll distribution in Tracadie Bay in 2000 showed reduced phytoplankton biomass within mussel leases consistent with the hypothesis of particle depletion and also showed interactions between leases (Fig. 3.1; E. Horne and G. Bugden, in preparation). Single mapping surveys with CASI and BIO-Acrobat have been successful at detecting variations in particle distributions that can be interpreted in the context of local variations in shellfish biomass. The ability of both these approaches to quantify particle depletion over bay-scales is greatly enhanced when the surveys are repeated several times over a tidal cycle. Coastal suspended particle concentrations, flux and geographic distribution all vary naturally over tidal-cycles due, in part, to the import and transfer of food through the system by tides. A time-series of particle distribution maps can be used to observe and quantify the depletion of imported food as it enters the bay and is transported from one lease to the next. Repeated BIO-Acrobat surveys of phytoplankton distribution in Tracadie Bay over spring- and neap-tidal cycles were conducted in 2003 and revealed large tidal-cycle variations in phytoplankton concentrations that are associated with a combination of tidal exchange with the Gulf of St. Lawrence, phytoplankton production in Winter Harbour fed by land-based nutrient enrichment, and particle depletion by the mussel culture (P. Cranford, in preparation; further details in Working Paper #4). These results showed the rapid depletion by cultured mussels of internal and external sources of food at the bay-scale, confirm the predictions of ecosystem and particle depletion models for Tracadie Bay (Dowd, 2005; Grant et al., in preparation; Working Paper #4), and help explain industry observations of relatively depressed mussel meat yields in this bay.

3.1.2. Particle depletion thresholds

Placing particle depletion measurements in the context of a potential HADD or net loss in habitat productivity is a requirement for the identification of regulatory decision thresholds. Unlike the benthic situation, the impact of shellfish aquaculture on the water column does not accumulate over time, but is relatively instantaneous and fluctuates in space and time with variations in hydrodynamic conditions. The extent of particle depletion is greatest within the lease footprint, but with seawater in near constant motion, depletion impacts are inherently a bay-wide issue, particularly in intensively leased mussel aquaculture areas. The production of organisms that feed on phytoplankton (secondary production) is often correlated with chlorophyll biomass or primary production. The introduction of invasive bivalve species to marine and aquatic environments has been shown to result in significant declines in crustacean zooplankton biomass, presumably related to enhanced food competition, and this reduced secondary production has led to declines in pelagic fish stocks (Cloern, 2005).

For shellfish culture to significantly impact pelagic habitat and productivity, particle depletion would have to: (1) be persistent; (2) occur over scales relevant to coastal ecosystems; and (3) be of a magnitude known to significantly depress natural populations of secondary producers. The first two criteria have been confirmed under conditions of intensive suspended mussel culture in PEI, while the latter remains speculative since no directed research has been conducted. Alternatives to setting a threshold based on the magnitude of particle depletion include setting an allowable zone of effect (e.g. no measurable depletion outside the lease footprint) or allowable biological exposure time (e.g. zooplankton exposure to depleted zone limited to proportion of average life-span).

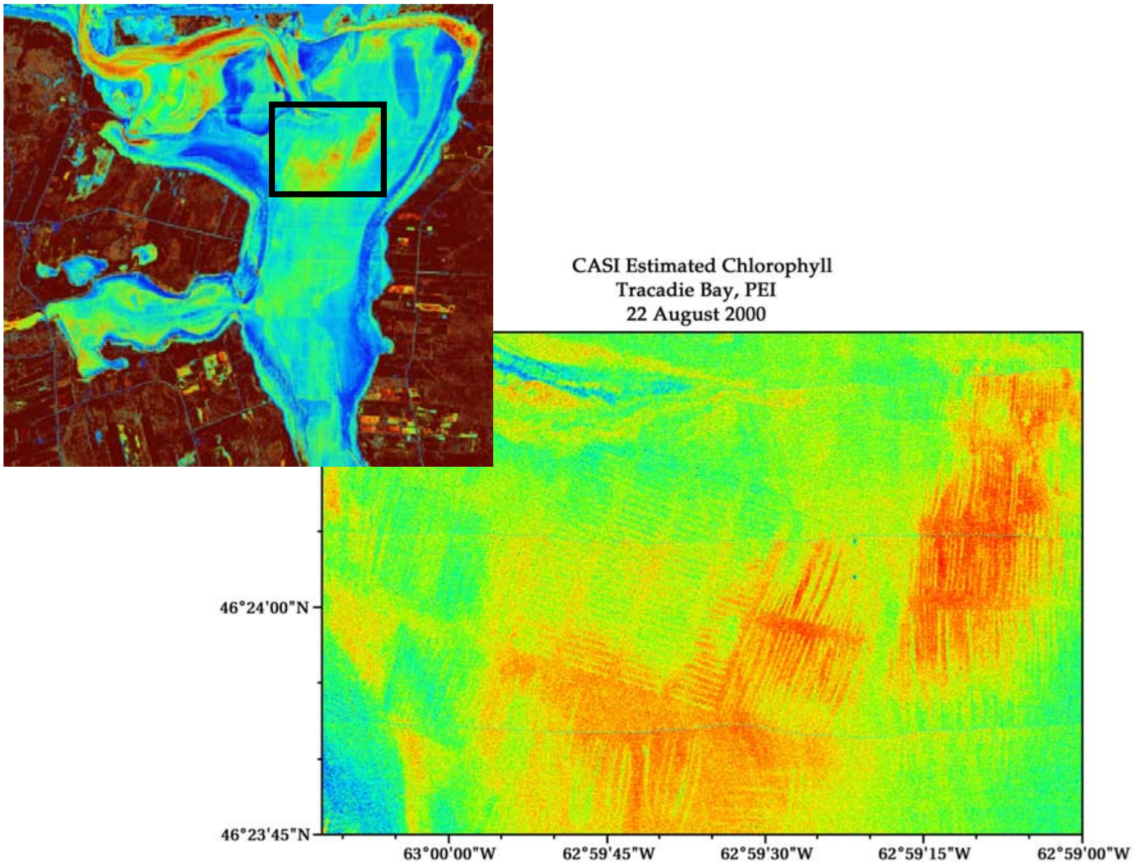


Figure 3.1. Estimated chlorophyll distribution in Tracadie Bay, PEI, on 22 August 2000, as determined using data collected by the Compact Airborne Spectrographic Imager (CASI). Relatively high chlorophyll shows as red, with decreasing values progressing from red to green to blue. The close-up image from the deeper central part of the bay, where the accuracy of the chlorophyll estimate is greatest, shows reduced chlorophyll levels in the vicinity of mussel lines. Water was moving from south to north at the time the image was taken.

Both may be of value for assessing impacts on secondary producers. The zooplankton are transported in and out of bays with the tide and competition for food with shellfish culture can be minimized by limiting the area of food depletion and/or the time of exposure. The latter could be estimated by scaling the embayment flushing time to zooplankton lifespan. A management threshold could be set based on relationships between food availability and zooplankton survivorship and/or reproductive output. However, owing to the general lack of research into the potential impacts of aquaculture on secondary production, the establishment of indicator thresholds for managing aquaculture-related particle depletion is not considered feasible at this time.

Coastal embayments are highly dynamic environments with a wide range of natural variation expected for all the identified indicators. No single measurement of any potential indicator (e.g. chlorophyll concentration) is capable of detecting particle depletion over scales relevant to a HADD. However, it has been shown that approaches are available, based on rapid large-scale surveys of indicator distributions that can quantify particle depletion reliably, consistently and objectively. Unfortunately, these approaches have either a high initial setup cost (instrument purchases) or require ongoing equipment rental (e.g. CASI). They also require specialized training to ensure proper sensor calibration and data interpretation. The only practical approach to the regional assessment of large-scale particle depletion at this time appears to be the establishment of partnerships between regulators and scientists towards the development of directed research programs designed to increase knowledge of aquaculture impacts on particulate food supplies and secondary producers, while also addressing some environmental management objectives.

3.2. *Oxygen* (G. Bugden)

3.2.1 *Introduction*

The concentration of dissolved oxygen (DO) in coastal waters varies on a variety of time and space scales. Oxygen exchange with the atmosphere, advective transport, photosynthesis, respiration, the chemical decomposition of organic matter and changes in water temperature, which affect the solubility of the gas, all play roles in determining the DO concentration. These processes vary on time scales ranging from diurnal to seasonal. The presence of suspended shellfish culture can affect the distribution of dissolved oxygen in several ways. The organisms themselves require oxygen for respiration and the flora and fauna attached to culture support structures may act as a source or sink for oxygen. Increases in the downward flux of organic matter brought about by the repackaging of suspended particles by filter-feeding shellfish can result in significant organic enrichment of the sediments beneath the culture operation, thereby increasing benthic oxygen demand (see Sections 2; Fig. 1.1).

A great deal of research has been conducted on the interactions between finfish aquaculture and the near field distributions of pelagic DO. Page et al. (2005) found that under conditions of high water temperatures and low wind speeds, both of which typically occur in the late summer and early fall in northern temperate regions, ambient DO concentrations could be reduced significantly within southwestern New Brunswick (SWNB) fish farms during intervals of reduced tidal currents. They also found that pelagic DO concentrations varied substantially on diurnal and semi-diurnal time scales, thus increasing the level of sampling required to monitor DO variations. Several other authors have developed models to estimate the concentration of DO within fish farms in SWNB (e.g. Trites and Petrie, 1995). Most of these models assumed that a balance between fish respiration and advective supply controlled DO concentration within the farms. The majority of these models did not explicitly include atmospheric or benthic fluxes or pelagic production/respiration.

Page et al. (2005) developed an oxygen depletion index as the ratio between the time required for fish respiration to deplete the DO within a farm to a chosen threshold level and the time needed to ventilate the farm. It was suggested that values of this index approaching 1 should trigger a more detailed examination of the DO budget for the farm under consideration. Although this approach may have some merit for finfish culture in the tidally dominated Bay of Fundy, it is not clear if it is directly transferable to shellfish culture.

Shellfish culture is often carried out in relatively sheltered areas with a high level of primary productivity that provides food for the shellfish. Vertical stratification of the water column is often significant and tidal exchange relatively small. A regional example is Tracadie Bay in PEI. Another is Etang de Thau in France, where intensive culture of oysters is conducted in a lagoon that has very limited tidal exchange. In the Thau Lagoon, episodic events of hypoxia, known locally under the generic name of “malaïgue”, have been responsible for major mortalities in both the natural ecosystem and the cultured stock (Mazouniet et al., 1996). During another Thau Lagoon observational program it was suggested that an anoxic event was narrowly averted by a period of strong winds that lead to a breakdown in vertical stratification and increased atmospheric flux (Plante-Cuny et al. 1998). Severe anoxic events affecting the culture operations in Tracadie Bay have not been observed but DO depletion near the bottom, presumably due to benthic demand, has been observed (Figs. 3.2 and 3.3). In Tracadie Bay a large fraction of the seabed is characterized by bacterial mats, indicating hypoxic-anoxic conditions at the sediment surface (see Section 2.3). Eh and sulfide measurements also indicate that sediments have a high oxygen demand related to shellfish biodeposits (see Section 2.1 and Working Paper #4). Benthic DO production would appear to be limited to around the edges of the Bay and the tidal flats near the mouth.

The flux of oxygen through the sea surface may be estimated from wind speed and water temperature (Stigebrandt, 1991). Using a wind speed of 5 m s^{-1}, a temperature of 20°C and the difference in DO concentration between the vertical average of the profiles shown in Figure 3.2 and the saturated concentration (based on surface water temperature and salinity), gives an estimated surface DO flux of 629 µM m^{-2} h^{-1}. Using the same DO concentration difference and an estimated e-folding flushing time of 81.6 h (Grant et al., 2005), gives an estimate of 633 µM m^{-2} h^{-1} for the surface flux, which assumes that offshore waters are at saturation. These estimates were combined with estimates of the other components of the DO budget for Tracadie Bay and the results are shown in Table 3.1. Although there are large uncertainties involved, the summer budget appears to be close to balanced. This indicates that the mussel aquaculture biomass in Tracadie Bay is potentially at an important threshold level, above which the dissolved oxygen demands would exceed the supply from natural processes.

In the winter, ice cover will limit wind mixing and atmospheric gas fluxes. Other components of the DO budget are also reduced during the winter, including the pelagic production. This may place an increased demand upon the tidal flushing to maintain oxic conditions. No observations of DO have been made through the ice at Tracadie Bay to examine the oxygen budget during the winter season. This is an observational gap that should be filled.

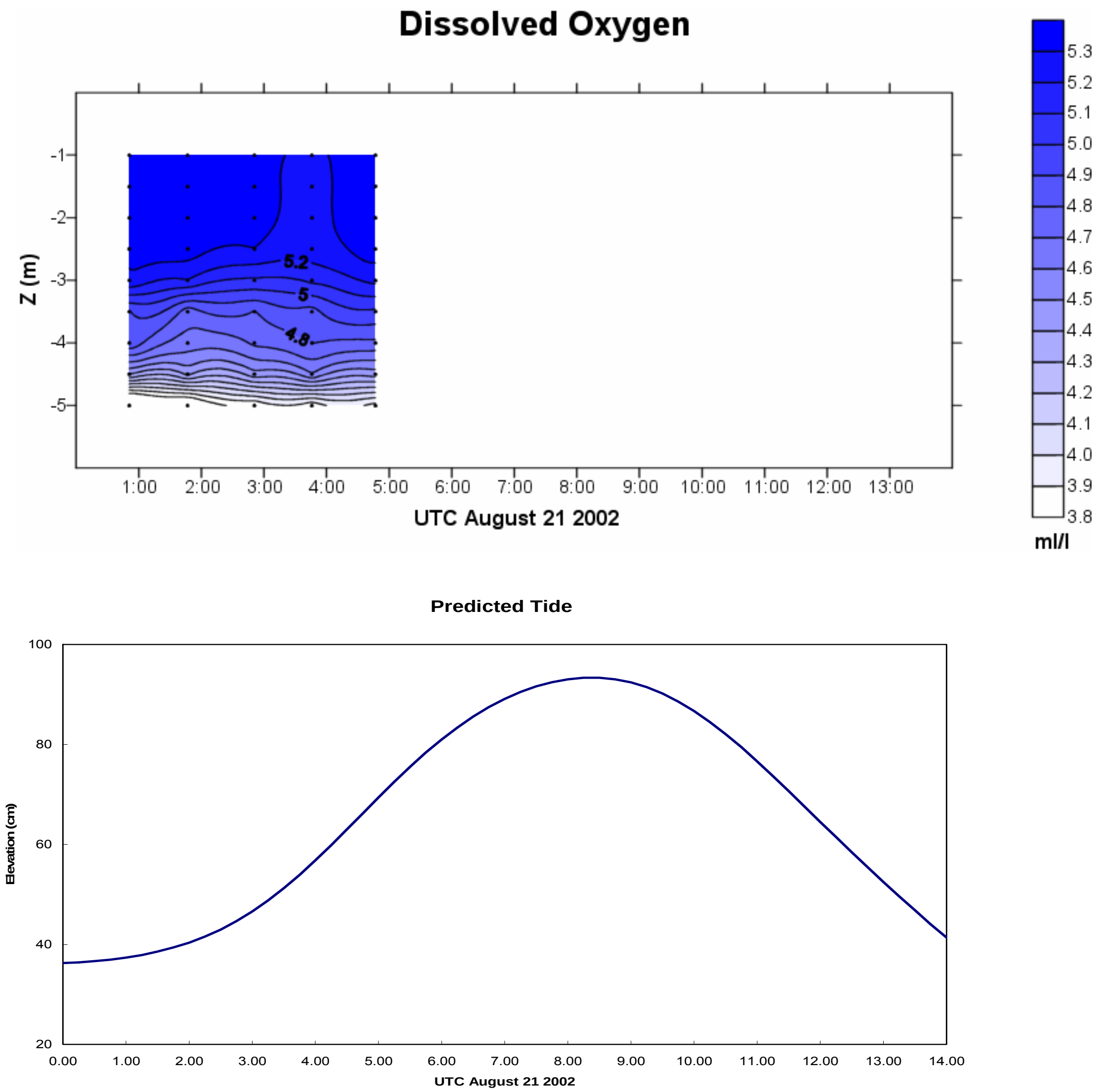
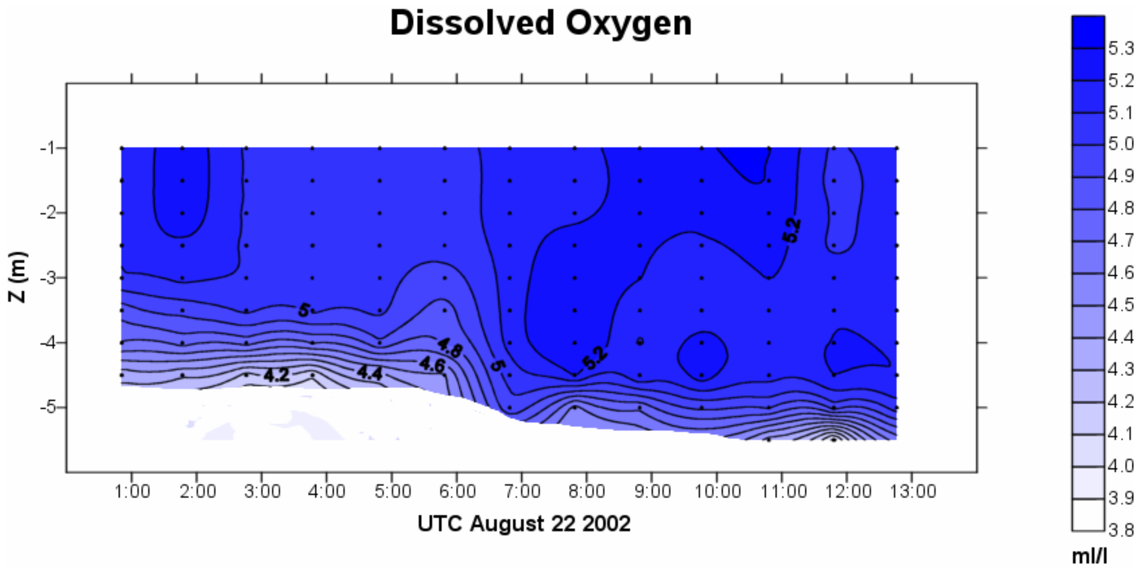


Figure 3.2. Time series of DO profiles from the central portion of Tracadie Bay, showing DO depletion near the bottom. It would take 75 hours to reach the observed profile from a uniformly saturated profile using observed benthic DO fluxes. This is very close to the calculated flushing time of 85 hours for the bay.



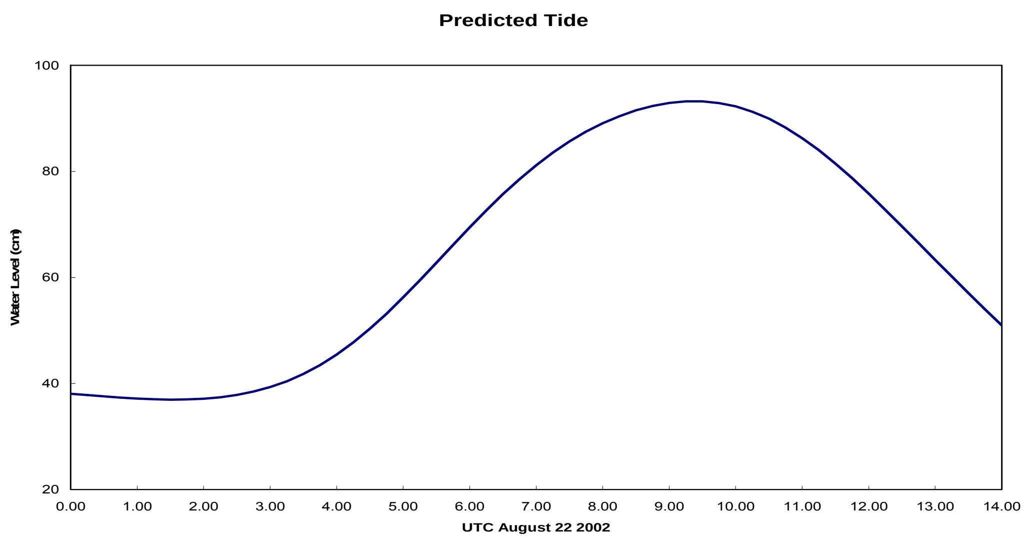


Figure 3.3. Time series of DO profiles from the central portion of Tracadie Bay, showing DO depletion near the bottom. Near surface DO values are enhanced at HW indicating that tidal ventilation may play a larger role than indicated by the observations of Page et al. (2002).

Table 3.1. Estimates of the various components of the DO budget for Tracadie Bay PEI. Fluxes are given as daily averages in (μM m^{-2} h^{-1}). Pelagic respiration is assumed to occur over the whole 4 m depth water column over a full 24 hours. Production is assumed to be limited to the upper 2 m for the 12 daylight hours.

	Summer	Winter	Source
Tidal Flushing	633	?	Calculated (see text)
Surface Flux	629	0	Calculated (see text)
Benthic Flux	-896	-896	B. Hargrave (pers. comm.)
Pelagic Respiration	-3000	-1720	P. Kepkay (Section 3.4.3)
Pelagic Production	4880	100	G. Harrison (Section 3.4.3)
Culture Respiration	-2027	-1161	P. Cranford (pers. comm.)
Total	219	?	

3.2.2 Discussion and recommendations

The individual components of the pelagic DO budget, such as respiration and production, vary diurnally and seasonally over a wide range. These variations are generally not resolved by sampling programs, thus making it difficult to recommend pelagic dissolved oxygen as an effective indicator of shellfish aquaculture effects. Approaches described for assessing oxygen status at finfish culture sites in the tidally dominated Bay of Fundy (see above) might not be transferable to shellfish, as shellfish culture is often carried out in locations where a balance between culture stock respiration and advective supply cannot be assumed. Shellfish culture is often carried out in locations that become ice covered during part of the year. This ice cover can limit gas and momentum exchange, which may result in reduced oxygen concentrations. This should be investigated further.

3.3 Nutrients as indicators of impacts from shellfish aquaculture (P. Strain)

As has already been noted, there is an extensive literature describing how high numbers of filter-feeding shellfish can alter nutrient dynamics in their environment (Working Paper #1; Dame, 1996; Cranford et al., 2003). Shellfish consume the organic nutrients in the organisms and detritus they feed on, excrete dissolved nutrients in their metabolic wastes, and deliver organic materials containing nutrients to the benthos in their feces and pseudofeces. The rate of nutrient cycling may be increased both by the excretion of dissolved wastes and by the more rapid remineralization of organic wastes in organically enriched environments, both in the community surrounding the cultured organisms and in the underlying benthos. Nutrients released by these processes may stimulate primary production. Shellfish cultured in coastal inlets also have the potential to trap nutrients in organic matter from both land and offshore sources. Nutrients are removed from the ecosystem by the harvesting of the shellfish.

High concentrations of shellfish can influence dissolved concentrations of inorganic forms of nitrogen, phosphorus and silicate: ammonia and phosphate are excreted, and ammonia, phosphate and silicate are released from benthic environments by the decomposition of shellfish biodeposits (Dame et al., 1991; Smaal and Prins, 1993; Prins and Smaal, 1994; Strain, 2002). Because nitrogen is usually considered to be the nutrient limiting primary production in coastal ecosystems, the most attention has been paid to nitrogen cycling. However, shellfish culture also has the potential to affect N:P and N:Si ratios, with possible consequences for phytoplankton dynamics (also see sections on Microbial Plankton Indicators [Section 3.5] and Harmful Algal Blooms [Section 3.4]).

As with all aquaculture impacts, a consideration of scale is crucial to any discussion of shellfish aquaculture impacts. Perturbations of nutrient dynamics inherently affect medium to large scales (hundreds of meters to tens of kilometres) and dissolved nutrients, whether derived directly from excretion or indirectly through the decomposition of organic wastes, are readily dispersed and/or transported over relatively large distances.

In examining the potential for nutrients to be useful indicators for the management of shellfish aquaculture, it is important to consider how the different processes might interact to produce observable changes in nutrient conditions. We have been asked to consider whether approaches to management of finfish aquaculture might also be applicable to shellfish aquaculture. A brief discussion of the nutrient impacts of finfish culture, whose management implications have already been assessed (Strain, 2005), provides a useful starting point for a discussion of shellfish impacts. Figure 3.2 displays the important pathways for nutrient discharges from net-pen finfish aquaculture. From a perspective of nutrient impacts, the fact that all the energy and nutrients required for the growth of the fish are derived from feed added to the environment is critical. A substantial fraction (e.g. ~60% of the nitrogen; Strain, 2005) of the nutrients added in the feed end up as wastes discharged to the environment. The fish do not depend on the natural marine environment for their feed, only for oxygen and a medium to remove their metabolic wastes.

The feed used in finfish aquaculture is an external, anthropogenic source of nutrients. In other words, the associated nutrient impact is a eutrophication issue. Although a number of different schemes have been developed to classify eutrophication conditions in nearshore waters, for the most part they depend on eutrophication symptoms and lack universal applicability because of the many different ways eutrophication can be manifested in the environment (Strain, 2005). Measures of nutrient concentrations (e.g. maximum winter nitrate concentrations) are part of some such schemes, but the application of direct nutrient measurements is complicated by the high natural variability of nutrient levels in both space and time. While widespread elevated nutrient levels have been measured or predicted in areas of intensive finfish farming (e.g. Bugden et al., 2001; Strain and Hargrave, 2005), setting meaningful management thresholds for inorganic nutrient concentrations, or attributing elevated nutrient levels to individual farms, is usually not possible. Furthermore, the addition of nutrients in wastes may not always translate into higher nutrient concentrations in the water column if those nutrients are quickly absorbed by primary producers or transferred to sediments. If enough supporting

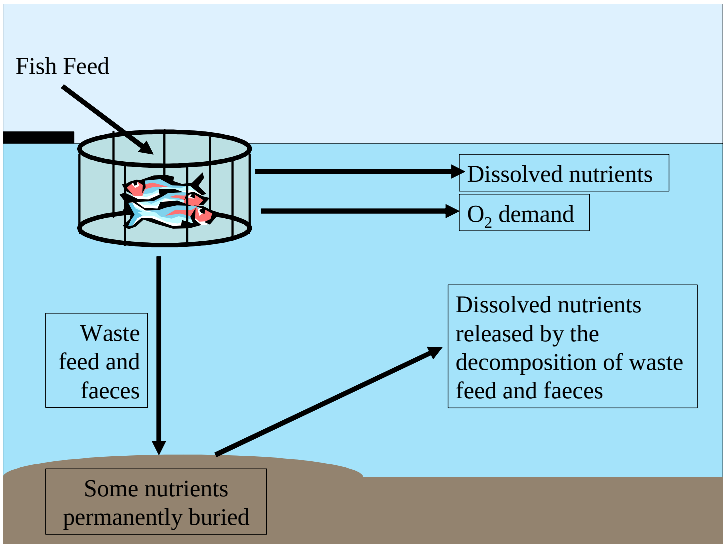


Figure 3.2. Principal pathways of nutrient impacts from finfish aquaculture.

data are available, however, measurements of nutrient levels or nutrient ratios might reveal changes in average nutrient levels, limiting nutrients (e.g. nitrogen:phosphate or nitrogen:silicate ratios), or the amount of new versus recycled production (ammonia:nitrate ratios). Comparative measurements of uptake rates (such as a nitrate/ammonia preference index) may also reveal changes in important aspects of ecosystem functioning. Because i) trends in any such measurements will take time to develop; ii) the impacts of any changes may be confounded by other important factors such as light that may limit growth; and iii) thresholds are not available, these measurements fall into the surveillance category of indicators described earlier, and are only warranted in areas where risk from aquaculture activity is high.

The magnitude of nutrient inputs from finfish aquaculture scale directly to the total number and size (or biomass) of fish in an inlet, making total fish numbers or fish biomass useful proxy indicators for nutrient impacts. The relationship between fish numbers and waste discharges can either be determined through industry records of fish growth and feed use, or by mass balance techniques that combine data on fish numbers, fish nutrition and estimates of feeding efficiency to predict discharges (Strain, 2005). Combining such predictions with some characteristics of the receiving environment (such as natural levels of productivity and inlet flushing rates) makes it possible to set thresholds for fish numbers (DFO, 2003; Strain and Hargrave, 2005).

Figure 3.3 shows the important pathways through which shellfish aquaculture can impact nutrient dynamics. Unlike the finfish case, in shellfish aquaculture as it is practiced in Canada, the shellfish are not fed, but rely on natural sources of food (phytoplankton and detritus) in the environment. This is an extremely important distinction from a nutrient perspective: the cultured shellfish are much more intricately linked to the ecosystem machinery than cultured finfish. The ability of bivalves to influence such fundamental properties as phytoplankton abundance, water clarity and

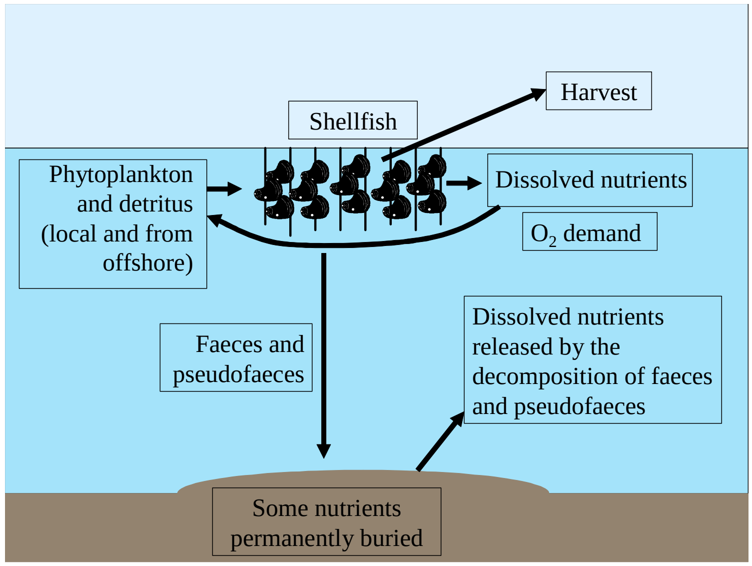


Figure 3.3. Principal pathways of nutrient impacts from shellfish aquaculture.

nutrient dynamics has led to their characterization as "ecosystem engineers" (Jones et al., 1994; Cranford et al., 2003). There are many potential feedback mechanisms between the shellfish, inorganic nutrients, and phytoplankton dynamics: (1) inorganic nutrients (excreted by the shellfish or formed by the decomposition of feces and pseudofeces) can promote phytoplankton growth; (2) the reduction of suspended particulate matter concentrations can increase light levels and promote phytoplankton growth; (3) competition between shellfish and zooplankton for food can influence phytoplankton bloom dynamics, etc. Again in sharp contrast to the finfish case, harvesting of cultured shellfish represents a net removal of nutrients from the marine environment. In that sense, shellfish aquaculture cannot cause a net eutrophication. However, organically enriched sediments under shellfish aquaculture sites are clear evidence that shellfish can redistribute nutrients, and in particular may focus nutrients in nearshore benthic environments causing local, sometimes severe, eutrophication (e.g. Dahlback and Gunnarsson, 1981; Kautsky and Evans, 1987; Hatcher et al., 1994). A nitrogen budget currently under development for mussel aquaculture in Tracadie Bay, PEI, suggests that the mussels direct ~15 times more nitrogen to the benthos than is removed by the harvest.

Although the enumeration of the processes through which high concentrations of shellfish can influence nutrient dynamics is straightforward, and field studies have confirmed that these mechanisms operate, predicting the net impact of these processes on nutrient levels is problematic. In writing about intertidal shellfish aquaculture in Baynes Sound on the east coast of Vancouver Island, Jamison et al. (2001) stated "The impacts ... of culture ... on community structure and ecosystem functioning (e.g. nutrient dynamics) are unknown and with available knowledge cannot easily be predicted." For some locations, different types of ecosystem models are available and are starting to give us a picture of how intense shellfish aquaculture can modify whole ecosystems (see Working Paper #2). Such models contribute significantly to our understanding of interactions between cultured shellfish and the environment, and will inform shellfish aquaculture management. But they are not yet capable of suggesting attributes of the nutrient cycle that would be suitable for monitoring, or thresholds of such variables that could be used to trigger management actions.

As there is for finfish, there may still be a role for direct measurements of nutrient levels or nutrient ratios in surveillance monitoring of shellfish aquaculture impacts; nutrient measurements are included in some environmental monitoring programs for shellfish aquaculture. For example, ammonia analysis is a required component of the environmental monitoring plan for St. Ann's Bay, NS, and nutrients are a recommended component in surveillance monitoring for shellfish aquaculture in Victoria, Australia (Gavine and McKinnon, 2002). The inclusion of nutrient measurements in such programs is a reflection of the potential for shellfish to alter nutrient dynamics, but often there is no clear statement of the purpose of the measurements or a plan of how the nutrient results will be interpreted and used. To be meaningful, nutrient sampling may have to be extensive, particularly when nutrient levels have a high local variability in time and space. Figure 3.4 illustrates the potential severity of this problem. The figure shows the results of nutrient measurements made at a fixed site at different stages of the tide over a two day period in Tracadie Bay, PEI. Ammonia, phosphate and silicate levels in the near-bottom samples varied by factors of between 3 and 4, depending on the stage of the tide. Considering all these factors, nutrient monitoring is speculative in nature and definitely falls into the category of surveillance rather than operational monitoring. Such monitoring should only be undertaken if the risk from shellfish aquaculture is thought to be high due to a high density of farms or high sensitivity of the local environment.

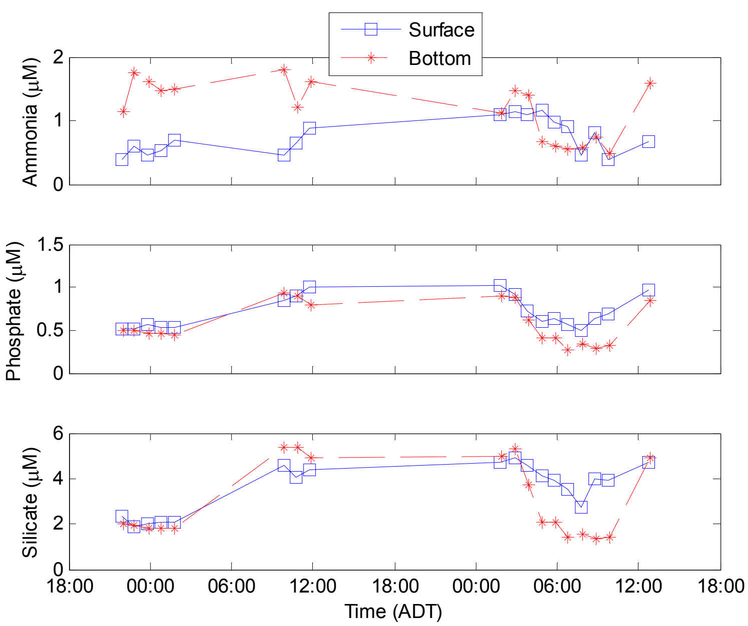


Figure 3.4. Nutrient concentrations at a fixed site in Tracadie Bay, PEI, taken at various stages of the tide on August 20-22, 2002. Samples were collected at depths of 1 m (surface) and 1 m from the bottom (mean water depth ~4 m) (P. Strain, unpublished data).

If direct measurements of nutrient concentrations are unlikely to be useful indicators of shellfish aquaculture impacts on nutrient dynamics, are there any parameters that can serve as proxies? In the case of finfish, fish numbers and biomass are useful proxies for the intensity of nutrient impacts. Unfortunately, there is not a comparable direct connection between the density of shellfish culture and the severity of the nutrient impacts. Although the number of cultured shellfish will determine the extent to which the underlying nutrient dynamics have been altered, and will continue to be a critical management parameter, the impacts of shellfish culture on nutrient dynamics will probably be most easily observed indirectly through a suite of other indicators, including phytoplankton composition and abundance, macrophyte distributions (de Casabianca et al., 1997), and shellfish growth rates (Bacher et al., 1991), that broadly relate to the structure and function of the ecosystem, and are discussed in other sections of this paper.

3.4 Microbial plankton indicators (W.K.W. Li, W.G. Harrison & P.E. Kepkay)

3.4.1 Phytoplankton abundance

Phytoplankton populations can be controlled by bottom-up factors (inorganic nutrients, light) and by top-down factors (grazing, viral lysis). In shellfish aquaculture, filter-feeding activity by shellfish has the potential to exert top-down control to an extent that is normally not present in natural ecosystems. At the same time, shellfish are able to increase inorganic nutrient availability in the water column by direct excretion and by supplying biodeposits for benthic mineralization. Therefore in aquaculture systems, there is both negative and positive feedback between the shellfish and the ecosystem. The net outcome of these interactions is contingent on specific conditions of the habitat, such as physical dynamics (tides, currents, wind) and nutrient loading (natural as well as anthropogenic sources). In general, grazing impacts are significant if the residence time of water is long, and if filter-feeding clearance rate is at least commensurate with phytoplankton growth rate.

Intense grazing of phytoplankton by shellfish has the potential to reduce photoautotrophic biomass, alter primary productivity, and change algal community composition (Prins et al., 1998). In these respects, a well-studied case is the Thau Lagoon on the Mediterranean coast, where oysters are farmed at levels of about 40,000 tons, among the highest in France. Here, filter feeding by oysters can lead to an average chlorophyll deficit of 44% compared to a reference site (Souchu et al., 2001). Thau Lagoon oysters preferentially retain particles larger than 5 μm, and therefore diatoms, dinoflagellates, phytoflagellates and to some extent ciliates constitute the major portion of their nutritional source (Dupuy et al., 2000). Particles less than 5 μm are less efficiently retained. Furthermore, the potential link from microbes to phagotrophic protists to shellfish is weak. Thus, small nanoplankton and all picoplankton (both photoautotrophic and heterotrophic) are neither consumed effectively by oysters nor transferred efficiently to them through trophic linkage (Dupuy et al., 2000). A more recent study in Thau Lagoon instead demonstrates a strong link from picoplankton to microzooplankton, and thence likely to mesozooplankton such as the appendicularian *Oikopleura dioica* (Bec et al., 2005).

Perhaps counter intuitively, specific primary productivity can be higher inside shellfish farming zones in spite of, or perhaps because of, intense grazing activities. In the Thau Lagoon, the primary production to chlorophyll ratio (PP:Chl) can be enhanced in the vicinity of shellfish farms because of nutrients made available from regeneration processes. In summer, high temperature becomes the primary determinant of the PP:Chl ratio, and therefore there is a diminished difference between farm and reference sites in this respect (Souchu et al., 2001).

The escape of picophytoplankton from oyster grazing is thought to be an important reason why large populations of such cells are found in the Thau Lagoon. Indeed, the smallest known eukaryotic picophytoplankter, *Ostreococcus tauri*, was discovered in this lagoon (Courties et al., 1994), constituting the most abundant component of the phytoplankton community. It is plausible that this particular species prevails over others in the same size class because of photoprotective pigments or possible heavy metal tolerance (Vaquer et al., 1996), which would be useful adaptations to life in shallow embayments within agricultural watersheds. In mesocosms, it has also been shown that mussel grazing can change phytoplankton species composition, shifting the community to one dominated by picoplankton forms (Prins et al., 1998), some of which (e.g. *O. tauri*) may have high intrinsic growth rates (Courties et al., 1998). The size spectrum of phytoplankton communities usually conforms to the general allometric law in which a power exponent of –3/4 prescribes the rate of abundance decrease with cell size (Li, 2002). A much steeper decline arising from an extraordinary abundance of picophytoplankton might be an ecosystem indicator of small size refuge in the face of intense macrofaunal grazing of microphytoplankton.

In Tracadie Bay, a site of intense mussel aquaculture in PEI, picophytoplankton are extraordinarily abundant (Fig. 3.5A). In fact, the abundance (10^{12} cells m^{-3}) exceeds that measured anywhere in open oligotrophic oceans where large populations of *Prochlorococcus* and *Synechococcus* exist (10^{11} cells m^{-3}). Unfarmed temperate coastal waters in eastern Canada (e.g. Bedford Basin; Scotian Shelf) and sites under the influence of finfish aquaculture (Bay of Fundy) also sustain high populations of picophytoplankton in the summer, but only on the order of 10^{11} cells m^{-3} (Harrison et al., 2005; Li et al., 2006). Picophytoplankton appear to be prevalent at other mussel cultivation sites in PEI as well. Smith et al. (1990) noted that 9% of the chlorophyll stock was in the picoplankton size fraction even near the height of a winter bloom in 1988 of the domoic acid containing diatom Nitzschia pungens in Cardigan Bay. The observation of 10-fold greater abundance of picophytoplankton around mussel farms in Tracadie Bay is consistent with the hypothesis of a small size refugia allowing these cells to gain a competitive advantage over phytoplankton species consumed by mussels. In unfarmed coastal waters, high chlorophyll concentrations are due largely to microphytoplankton such as diatoms and dinoflagellates. In Tracadie Bay, it appears that picophytoplankton also contribute substantially to the observed bulk phytoplankton biomass (chlorophyll *a*).

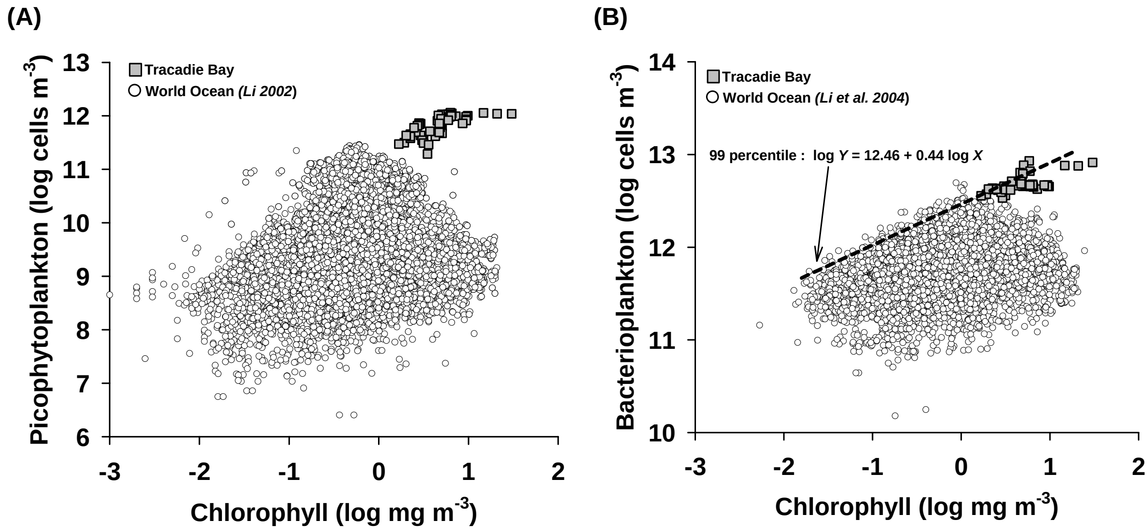


Figure 3.5. Extraordinarily high cell concentrations of picoplankton in Tracadie Bay, PEI, compared to worldwide oceanic and neritic waters. (A) Photoautotrophic picophytoplankton (B) Heterotrophic bacterioplankton.

3.4.2 Bacterioplankton abundance

Bacterioplankton populations are likewise controlled by abiotic (nutrients, temperature) and biotic factors (protistan grazing, viral lysis). However, unlike phytoplankton, which are photoautotrophic, bacterioplankton are heterotrophic and thus require preformed organic substrates. These substrates may be (1) labile photosynthates exuded by healthy phytoplankters; (2) egesta released by protozoans that have consumed phytoplankters; (3) cytoplasmic material liberated by viral lysis and algal autolysis; or (4) dissolved organic matter originating from shellfish biodeposits. Little is known about the interactions between cultured shellfish and bacterioplankton from direct research in farm regions. It is clear that sediment organic enrichment resulting from mussel biodeposition can lead to enhanced benthic bacterial abundances (Mirto et al., 2000). Yet, it does not necessarily follow that water column bacterioplankton will exhibit a commensurate response. As a case in point, the spatial and temporal structures of bacterioplankton production in the Thau Lagoon are likely controlled by wind-driven hydrodynamical processes for 50% of the time (Trousseillier et al., 1993).

In finfish aquaculture, an operation in which there is inadvertant excess input of organic matter (fish feed) into the ecosystem, it is possible to detect a spatial gradient of increasing bacterioplankton abundance towards the core of the farm (Sakami et al., 2003). At the mouth of the bay distant from the farm, bacterial production is supported by phytoplankton primary production. However, in the finfish aquaculture region, there is a positive anomaly in bacterial production, which can be attributed to heterotrophy supported by unconsumed feed and fish excreta (Sakami et al., 2003).

The Japanese example points to an approach for designating an ecosystem indicator of aquaculture impact. Bacterial abundance that is over and above the maximum level supportable by autochthonous primary producers can be ascribed to allochthonous sources. If the residence time of water in the embayment is sufficiently long, much of any positive bacterial anomaly might be ascribed to the presence of aquaculture. What then is the autochthonous carrying capacity for bacterioplankton?

Theoretical estimates of this carrying capacity based on the flux of utilizable organic matter from primary production and bacterial maintenance efficiency are from 4 to 8 x 10^{12} cells m^{-3} (Ducklow, 2001). Empirical measurements indicate that maximum bacterial abundance (Y) is related to chlorophyll concentration (X) as $\log Y = 12.46 + 0.44 \log X$ (Fig. 3.5B). This means that the bacteria to chlorophyll ratio decreases with increasing X. For X values of 1 and 10 mg Chl m^{-3}, predicted Y values are approximately 3 and 8 x 10^{12} bacteria m^{-3}. In Tracadie Bay, measured bacterial abundances are at levels essentially equivalent to carrying capacity at the given chlorophyll concentrations (Fig. 3.5B). This contrasts with oceanic conditions where bacterioplankton populations are reduced by top-down pressures when chlorophyll concentrations exceed about 1 mg m^{-3} (Fig. 3.5B and Li et al., 2004). It might thus appear that in Tracadie Bay, autochthonous phytoplankton may be sustaining bacterioplankton at maximum capacity, but the latter are not being checked by top-down control as they are elsewhere. This suggests a testable hypothesis: intense shellfish feeding of the natural bacterial grazers causes a trophic cascade allowing bacterioplankton to attain abundances as high as can be sustained by resident phytoplankton. If true, this would also imply that the trophic cascade also extends to picophytoplankton since they are on average only slightly larger than heterotrophic bacterioplankton. Moreover, the selective removal of larger phytoplankton and microheterotrophic grazers by cultured shellfish will impact other natural components of the foodweb for which the larger net-plankton are a primary food source (e.g. zooplankton, invertebrate larvae, etc.). Put another way, it is possible that overall system primary productivity at shellfish aquaculture sites can be sustained by small phytoplankton, in a compensatory way, but transfer of that energy through the food web will likely be altered with largely unknown consequences.

3.4.3 Production, respiration and the P/R ratio

One of the fundamental indicators of the flow of energy through an ecosystem is the balance between primary productivity and respiration in the water column. This balance is normally quantified by measures of the production and loss of dissolved oxygen. As mentioned earlier, coastal aquaculture sites are often subject to nutrient loading from a variety of natural and man-made sources (Strain and Hargrave, 2005). Despite this abundance of nutrients, enhanced phytoplankton biomass or productivity are not generally observed due to growth limitation from other factors, principally light (Cloern, 1999; Harrison et al., 2005). Tracadie Bay is a shallow and highly turbid system in which light limitation of primary productivity prevails, even for the smaller picoplankton. Despite light limitation, primary production in Tracadie can still attain high levels in summer (1-3 g C m^{-2} d^{-1}); winter levels are on the order of 0.05-0.1 g C m^{-2} d^{-1}. The production of bacteria, in contrast, is not constrained by light availability and may, in fact, be stimulated by shellfish release of organic substrates and by their predation on bacteria grazers (generating a trophic cascade). As the bacteria increase in number, respiration increases (Fig. 3.6) and dissolved oxygen (DO) will decrease. However, oxygen balance and ecosystem integrity will also be regulated by the production of oxygen by the phytoplankton (P) as well as bacterial respiration (R). The balance between these two processes can be expressed in terms of the P/R ratio (Kepkay et al., 2005). When P/R is greater than 1.0, an autotrophic ecosystem is in place and when the ratio is less than 1.0, oxygen demand by the heterotrophs is predominant. Oxygen production in Tracadie Bay exceeds demand by a wide margin in the summer, but demand exceeds production by an equally wide margin in the fall/winter (Fig. 3.7). When this large seasonal swing from oxygen production to demand is combined with the proliferation of bacteria, the overall increase in demand could force the ecosystem into a year-round heterotrophic state. This would be detected by high respiration and low P/R ratios that back up or carry over from fall/winter into the summer growing season (Kepkay et al., 2005).

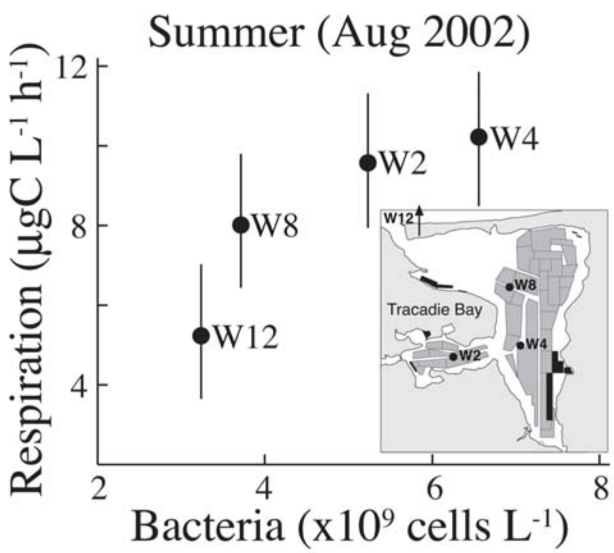


Figure 3.6. Correlation between water-column respiration and bacterial abundance in Tracadie Bay, summer 2002.

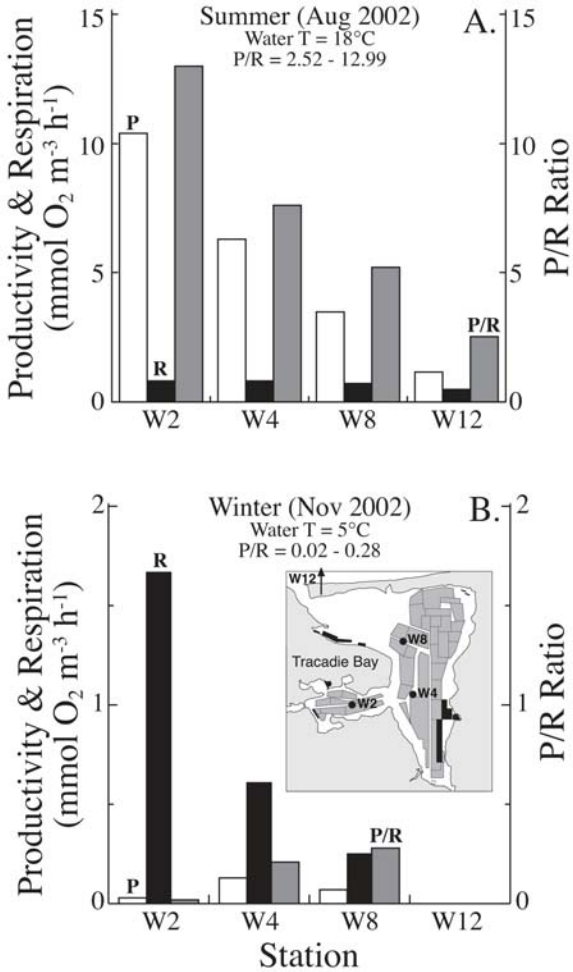


Figure 3.7. Primary production (P) and respiration (R) in Tracadie Bay indicating that P/R>1 in summer (A), and P/R<1 in winter (B).

3.4.4 Indicators to monitor

Observations made in Tracadie Bay and elsewhere suggest that some simple indicators of plankton community structure and metabolic activity might be relevant in assessing the effects of shellfish aquaculture on habitat. These would include: (1) seasonal tracking of the abundance of picophytoplankton (Pi) and bacterioplankton (B) for comparison with more conventional indicators of phytoplankton abundance (chlorophyll *a* concentration, CHL), in order to derive Pi/CHL and B/CHL ratios; and (2) seasonal tracking and targeted monitoring of primary production (P), microbial respiration (R) and the P/R ratio to aid in the interpretation of ecosystem trophic status. In the absence of direct measurements of primary production and microbial respiration, B/CHL ratios may serve as first-order indictors of P/R ratios (Harrison at al., 2005).

3.4.5 Performance indicators and thresholds

Because the ecological consequences of altering phytoplankton and bacterioplankton community structure or the balance between plankton production and respiration are complex and not well understood, it is difficult to define specific thresholds as a basis for decision making or management action. The operational strategies for managing human activities such as shellfish aquaculture are effected through so-called “performance indicators” and their respective thresholds or “reference points” (Gavaris et al., 2005). Thus, performance indicators reflect the ecosystem response of regulating human activities and the reference points or thresholds determine the point at which intervention (management action) is required.

Studies employing an ecosystem-based management approach include a suite of potentially relevant ecosystem traits that are considered to be important to monitor but for which specific performance indicators and thresholds are, at present, poorly or completely undefined. This is due in large part to a limited understanding of ecosystem structure and function. The tracking of ecosystem traits that are thought to affect productivity, community structure and habitat, (referred to as “contextual indicators” by Gavaris et al., 2005), is warranted. The indicators of phytoplankton/bacterioplankton community structure and primary production/respiration described above fall into this contextual category.

3.5. Harmful algal blooms (S. Bates)

Harmful algal blooms (HABs) in Canadian waters, as elsewhere, most commonly negatively impact aquaculture by causing mortality of net-pen finfish (e.g. Haigh and Taylor, 1990; Yang and Albright, 1994) and by contaminating molluscan shellfish with phycotoxins (e.g. Cembella and Todd, 1993; Bates, 1997; Bates et al., 1998). Thus far, there is no compelling evidence for the opposite effect; i.e. that aquaculture results in any significant impact, positive or negative, on the incidence of HABs. Finfish aquaculture has most often been implicated in causing HABs, or in making HAB species more toxic. However, there is little hard data to support this. For example, concerns were raised, most often by opponents of finfish aquaculture, that the discharge of nutrients from fish farms in Scottish coastal waters could be linked to increases in HABs (Staniford, 2003). The Scottish Executive has commissioned several reports on the subject, all of which have concluded that there is no good evidence linking the growth of fish farming with harmful algae (e.g. Tett and Edwards, 2002; Rydberg and Stigebrandt, 2003). A similar conclusion was reached for salmon farming in the Bay of Fundy (Martin and LeGresley, 1998). The ICES Working Group on Environmental Interactions of Mariculture (ICES, 2000) concluded that “in general, salmon farming activities did not lead to a detectable increase in nutrient concentrations.” Indeed, the aquaculture of molluscan shellfish differs substantially from that of finfish, in that food pellets are added to pens where the fish are raised, whereas molluscs filter feed on the natural particles already present in the water column (e.g. Shumway et al. 2003; see Section 3.3). One would therefore expect important differences in potential impacts between the aquaculture of these two types of animals. This section focuses on molluscan shellfish aquaculture with respect to its possible influence on HABs. It also addresses the question of which indicators and thresholds, if any, the monitoring of HABs may provide to address the issue of "harmful" alterations to fish habitat.

Historically, PEI embayments where mussel aquaculture occurs have been impacted mainly by HABs composed of the diatom genus *Pseudo-nitzschia*, some species of which produce domoic acid, the neurotoxin that causes amnesic shellfish poisoning (reviewed by Bates 1998, 2004; Bates et al. 1998). The first major outbreak was in Cardigan Bay during the late autumn and early winter of 1987, which caused the death of at least three people and the sickness of over 100. The toxin source was identified as *Pseudo-nitzschia multiseries*; this was the first instance of a diatom producing a phycotoxin. Toxic blooms in the subsequent two years were smaller, and the next closures were later, on the north shore: New London Bay (1991, 1992, 1994), Malpeque Bay(1991, 2001), and Mill River (2000). Major blooms of the non-toxic *Pseudo-nitzschia calliantha* (previously called *P. pseudodelicatissima*) occurred during 2001-2002. In the spring of 2002, most of the southern Gulf of St. Lawrence, including northern PEI, was closed due to domoic acid, traced to *P. seriata*. It is not known why there have been only minor blooms of the non-toxic *P. pungens* thereafter. Western and southwestern portions of PEI were closed due to paralytic shellfish poisoning (PSP) toxins, produced by the dinoflagellate *Alexandrium tamarense*, in the summer of 2003. In the autumns of 2001 and 2003, the dinoflagellate *Karenia mikimotoi*, which potentially kills fish and benthic invertebrates including mussels, was found at high concentrations in Cardigan Bay. Each of these HAB species was found at PEI mussel aquaculture sites, yet there is so far no evidence to suggest that their presence was related to aquaculture activities.

A survey of the literature shows no studies that address, specifically, the possible impacts of molluscan shellfish aquaculture on the incidence or toxicity of HABs in Canada, or elsewhere in the world. The literature does document the possible cumulative effects of green-lipped mussel farming (*Perna canaliculus*) farming on some aspects of the marine environment, mainly seabirds and mammals, in New Zealand (Lloyd, 2003; Butler, 2003). The main difficulty in making any link between aquaculture and HABs lies in the complexity of factors responsible for generating HABs, and in our inability to understand adequately all of these factors. In addition, there are factors that influence HABs other than those originating directly from aquaculture activities (e.g. nutrient inputs from sources other than aquaculture, fluctuations in light and temperature, the presence of natural populations of shellfish, etc.). It is presently a challenge to distinguish the origin and relative importance of these factors. The shellfish aquaculture industry and HAB monitoring are both relatively new endeavours. Until longer-term monitoring programs are in place, it will be difficult to identify any links between aquaculture and HABs that may exist. For these reasons, it is presently unrealistic and premature to expect the monitoring of HAB species to provide any useful indicators or indices for the management of shellfish aquaculture.

A useful example of the difficulties of using phytoplankton as indicators is work done by the ICES Study Group to Review Ecological Quality Objectives for Eutrophication, which examined the use of phytoplankton assemblages as indicators of nutrient enrichment (ICES, 2004). The Study Group concluded that the use of individual phytoplankton species or groups as environmental indicators has been very unsuccessful. It also expressed concern over the use of harmful and nuisance species as direct responders to elevated nutrients and found no convincing evidence that HAB events are nutrient enrichment driven events, either generally or for individual HAB species. Furthermore, they concluded that HABs do not generally have the desired properties as indicator species, i.e. some HAB species rarely form dense blooms and they occur systematically also in nutrient-poor areas. Nutrients are but one factor that may control phytoplankton growth dynamics and community composition, and HAB species are only an occasional subset of a phytoplankton community. Therefore, it is even more of a challenge to try to link shellfish aquaculture activities to any HAB species.

Nevertheless, as a start, the following factors may be important when considering how shellfish aquaculture may influence phytoplankton in general and HAB species in particular: (1) filtration by the molluscan shellfish; (2) release of nutrients by the shellfish; and (3) possible introduction of HAB species during the transfer of shellfish aquaculture products; and provision of habitat to HAB species.

3.5.1. Filtration by molluscan shellfish

Filter-feeding mussels graze on the phytoplankton in the water column, filtering from one to four litres per hour per animal (e.g. Rice, 2001). One consequence of this is to decrease the turbidity of the water. The resulting increase in available light could favour the growth of all remaining phytoplankton, providing an advantage for the mussels. However, the growth of any HAB species present could also be stimulated. So far, there is no evidence that any HAB species could take advantage of this increased light to outcompete any other species.

Grazing by mussels can also alter the community composition of phytoplankton by selecting for or against HAB species. For example, Rice (2001) describes a mesocosm experiment designed to determine the effects that aquacultured oysters have on the environment. It was found that diatoms of the genus *Nitzschia* (now called *Pseudo-nitzschia*) were predominant in the mesocosms with oysters, whereas *Skeletonema* was dominant in the control tanks. However, it should be noted that deposits from filter-feeding activity can also result in increased nutrient cycling from the sediments, thus promoting the growth of certain phytoplankton, as described below.

3.5.2 Release of nutrients by shellfish

Some of the potential impacts due to the release of nutrients by aquacultured shellfish are discussed in Section 3.3.2. From the point of view of phytoplankton, and HAB species specifically, such nutrients may: (1) stimulate primary production and biomass (provided that no other factors, e.g. light, trace metals, grazing, are limiting); (2) select for toxic species; and (3) increase the toxicity of some algal species. No studies were found that could provide any evidence linking nutrients from shellfish aquaculture to the promotion or toxicity of HABs. Again, for finfish aquaculture, the Scottish Executive

(2002) concluded that “except perhaps in a few enclosed waters, enrichment by fish farm nutrients is too little, relative to natural levels” to have caused an increase in the incidence and toxicity of HABs. This conclusion was tempered, however, by the lack of long-term monitoring programs needed to derive strong correlations.

The mechanism by which nutrients may select for certain phytoplankton, including toxic and harmful species, is embodied in the “nutrient ratio hypothesis” (Smayda, 1990). It is based on the differential requirement by some groups of phytoplankton for the elements N, P and Si. For example, diatoms require silicon in their cell walls, whereas most other phytoplankton do not. Thus, when the ratios of Si:N or Si:P in coastal waters decrease, diatom growth in these waters will generally cease, and flagellates may become more dominant. An exception, however, appears to be the toxic diatom, *Pseudo-nitzschia multiseries*, which can outcompete other phytoplankton at low Si:N ratios (Sommer, 1994). A preliminary analysis of nutrient ratios in PEI embayments shows that the Si:N ratio is frequently less than one during the autumn, suggesting Si limitation (P. Strain and S. Bates, unpublished data) and indicating favourable conditions for *Pseudo-nitzschia* spp. growth.

The toxicity of HAB species may be affected by which nutrient is limiting, or is in excess. For example, domoic acid production by *P. multiseries* is increased when Si or P becomes limiting (Bates, 1998). Thus, the low Si:N ratios that may be found in PEI embayments, in addition to promoting *Pseudo-nitzschia* growth, may also favour the production of this toxin. Limitation by P increases the toxicity of *Alexandrium* spp. that produce PSP toxins (Anderson et al., 1990). Finally, an excess of ammonia has been found to increase the toxicity of *P. multiseries* (Bates et al., 1993). This laboratory result cannot easily be extrapolated to the field situation, even though both finfish and shellfish aquaculture are sources of ammonia. For example, it has been used as an argument, disputed by the Scottish Executive, that ammonia from Scottish fish farms has promoted the growth and increased the toxicity of *Pseudo-nitzschia* spp. in their waters (Allan Berry, personal communication)

3.5.3 Possible introduction of HAB species during the transfer of aquaculture products

HAB species may be transferred from one embayment to another during transfer of product (e.g. spat). The PEI shellfish aquaculture industry is aware of this issue and has raised related questions when applying for permits to transfer product. However, the DFO Introductions and Transfers Committee has not yet established a policy to deal with this issue. The PEI Aquaculture Alliance is striving to rectify this for eastern Canada (Crystal McDonald, personal communication). Decisions regarding transfer permits are hampered because of a general lack of knowledge of the current distribution of HAB species in our waters. A comprehensive phytoplankton monitoring program would help to alleviate the problem.

3.5.4 Provision of habitat for HAB species

Virtually all HAB species are pelagic, with the exception of the benthic dinoflagellate *Prorocentrum lima*, which may also be found growing on the surface of aquacultured mussels, associated gear, and on vegetation attached to the gear. This was the source of the diarrhetic shellfish poisoning (DSP) toxins found in aquacultured mussels in Mahone Bay, NS (Lawrence et al., 1998) and in the southern Gulf of St. Lawrence. This is the only known Canadian HAB species that has such a habitat, but indicates the potential problem that could be caused by gear used in shellfish aquaculture.

3.5.5 Indicators and thresholds for managing shellfish aquaculture

Silvert (2001) is the only author found who has considered the theoretical risks of aquaculture to HABs, and the approaches by which this could be assessed. As well, he concluded that "there is no clear evidence as yet that aquaculture has led to greater incidence of toxic algae (e.g., red tides), but this is certainly a risk that has to be taken into consideration." Silvert (2001) outlined three basic parts to computing the effects of aquaculture on HABs: "the first is to calculate the change in nutrient and light levels due to aquaculture. The second is to estimate how these changes will affect primary production. The third is to identify whether any additional primary production will be harmful or not." Each of these steps is a great scientific challenge, especially the third, because of the complexity of ecosystems and factors other than aquaculture that must be considered.

Given that we do not yet understand all of the factors that promote and control HABs, it is currently not possible to use HAB monitoring as a source of any indices or indicators that could be applied to protect fish habitat from the impacts of shellfish aquaculture. However, the surveillance monitoring (as defined in Section 1.2) of phytoplankton assemblages associated with aquaculture sites, along with relevant chemical and physical parameters, *might* one day tell us if aquaculture has seriously disrupted the ecosystem. A similar conclusion was reached in the case of nutrients (Section 3.3). In the short term, the surveillance monitoring of HABs will provide information on the nutritional quality of the available food, as well as on the presence of any toxic or harmful phytoplankton that could negatively impact the shellfish product. In the longer term, it will help to build a better understanding of factors, including the possible influence of shellfish aquaculture, that lead to the incidence of HABs. Potential indicators to consider that could be linked to shellfish aquaculture include: changes in nutrient ratios, ratios between functional groups of phytoplankton (e.g. diatoms/dinoflagellates); and species succession leading up to the appearance of HAB species, perhaps in relation to neighbouring reference sites. Similar to the ICES Study Group aimed at assessing eutrophication impacts (ICES, 2004), any potential threshold should be determined in relation to natural variation at each site, and the action levels should deviate consistently from reference data, as well as persist over time. Until we achieve a better understanding of control factors for HAB dynamics, we will be unable to apply the appropriate indicators or indices for the managing shellfish aquaculture.

4.0 FARMING ACTIVITIES AND SHELLFISH PERFORMANCE

(L.A. Comeau)

4.1. Farming activities

As described in the introductory section of this document, intensive shellfish farming may result in the harmful alteration, disruption or destruction (HADD) of fish habitat. Accordingly, there is a requirement for establishing indicators that could signal a potential HADD. Almost all the indicators under considerations in this document are aimed at gauging the “effect” side of the equation (e.g. pelagic particle depletion, benthic organic enrichment). Here we consider for a moment the potential value of monitoring the causal factor, shellfish farming activities.

Historically, the monitoring of shellfish farming activities has been driven by the desire to maximize production with minimal attention directed towards issues of carrying capacity. Such monitoring has, however, led to the recognition of stocking limits, namely in Japan (9,000 oyster rafts in Hiroshima Bay; Figure 4.1), France (250 mussel “bouchots” per 100 m and 6,000 oyster bags per hectare; Kopp, 2001), Spain (500 ropes per mussel raft, Fuentes et al. 2000), and more recently Canada (500 mussel socks per acre in Tracadie Bay PEI; Lea 2002). Typically, stocking limits are only established by the industry and regulatory agencies following noticeable reductions in commercial yield.

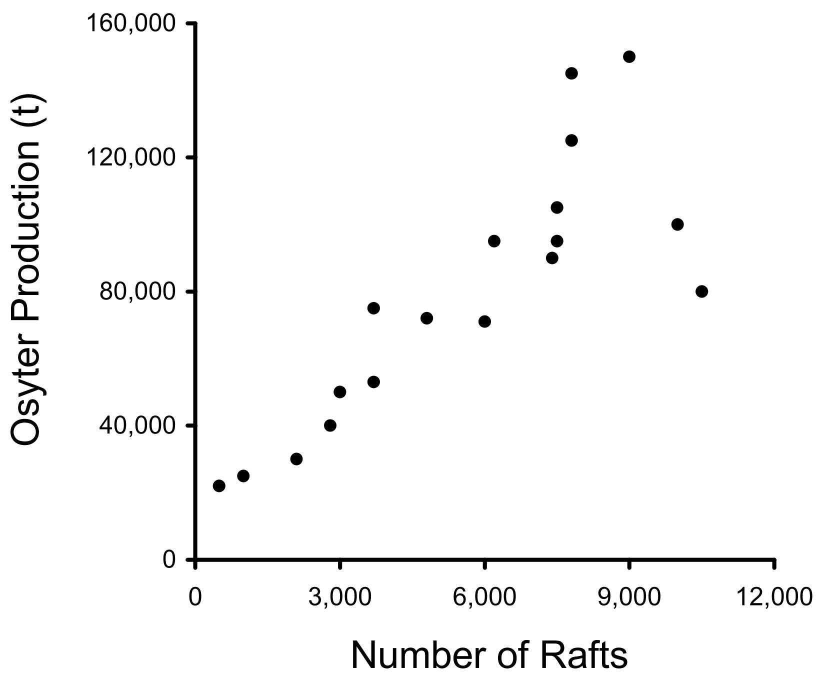
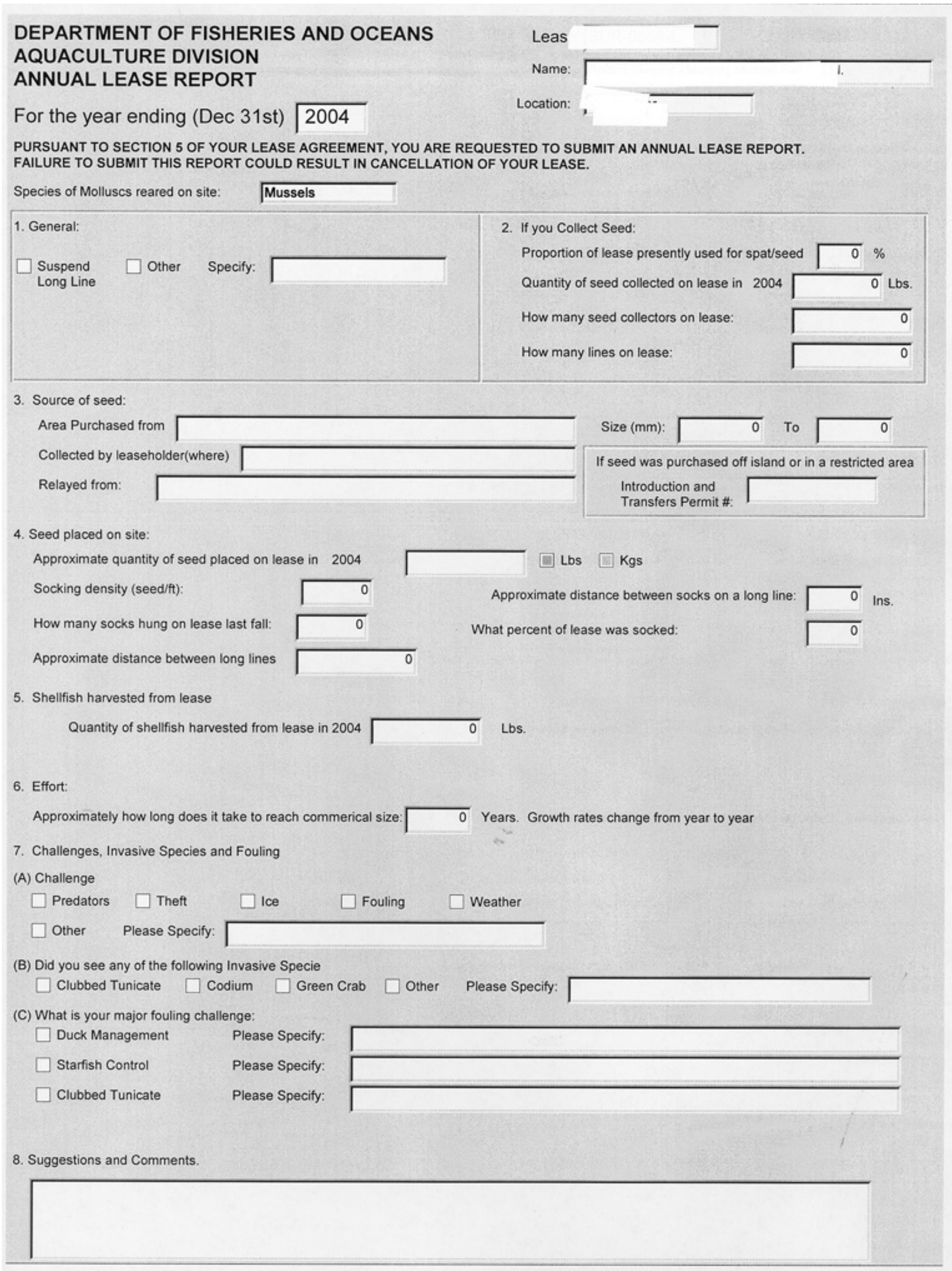


Figure 4.1. Annual production of oysters (1953 to 1970) in relation to the number of culture rafts in Hiroshima Bay, Japan. Adapted from Mallet and Myrand (1995).

A more defensible management approach from an environmental perspective would consist of setting farming activity thresholds based on a potential HADD of fish habitat. But the obvious difficulty with this proposition is that the science is not sufficiently advanced to recommend activity thresholds. It is reasonable to assume, however, that a sustained monitoring of farming activities (e.g. shellfish biomass) in concert with impact indicators (e.g. benthic organic enrichment) would lead to a better understanding of the putative cause-effect relationships, and perhaps ultimately to the establishment of activity thresholds. For instance, stock inventory thresholds could be determined using computer models based on HADD-avoidance measures and local oceanographic conditions. This line of reasoning relates to one of the EEM objectives listed under Section 1.2 (j - "to understand and delineate cause-effect relationships").

A few recent initiatives in Atlantic Canada have incorporated a number of husbandry parameters into their monitoring protocol. These variables are listed under the "St. Ann's Harbour Environmental Management Plan" (Stuart, 2003). Similar variables are found in the PEI "Annual Lease Report" shown below in Figure 4.2. Efforts are being made to collect information on the extent of biofouling, particularly in respect to fouling species that compete for available food resources, and also on the degree of product fall-off. However, the most important variables are those that relate to sleeving and seed deployment and harvesting activities. This information is essential for computing and keeping track of the total biomass. For instance, sleeving and harvesting data provided by leaseholders in Tracadie Bay PEI allowed a back-calculation of the historical development of mussel culture in that bay. Total mussel biomass under cultivation ranged from 1,100 t (1990 annual mean) to 4,700 t (2001 annual mean). Within each year, there are remarkable seasonal patterns in total biomass, which is typical of any mussel-producing embayment. Figure 4.3 shows that the total biomass generally declines from July to September due to harvesting activities. In contrast, the total biomass peaked in November-December following the onset of sleeving and seed deployment. These changes in shellfish inventory were important both in terms of the magnitude and the rate at which they occurred. In 1998, for instance, the total biomass increased by 37% between the months of September and December. The example of Tracadie Bay underscores the importance of monitoring inventory on a long-term basis.

Understandably, any monitoring of inventory raises concerns about privacy, since shellfish stocks are regarded as private business information. In 2000, the Atlantic Veterinary College interviewed Tracadie Bay, PEI, growers on that issue. A striking majority of growers were favorable to divulging inventory data on a periodic basis, recognizing that such a dataset could further the understanding of the embayment, particularly with regards to carrying capacity (economically speaking). The only concern expressed was one relating to confidentiality, and more specifically that any monitoring program include legal confidentially agreements, so that the data would not be divulged publicly in way that it could be traced back to individuals.



DEPARTMENT OF FISHERIES AND OCEANS
AQUACULTURE DIVISION
ANNUAL LEASE REPORT

Leas

Name: I.

Location:

For the year ending (Dec 31st) 2004

PURSUANT TO SECTION 5 OF YOUR LEASE AGREEMENT, YOU ARE REQUESTED TO SUBMIT AN ANNUAL LEASE REPORT. FAILURE TO SUBMIT THIS REPORT COULD RESULT IN CANCELLATION OF YOUR LEASE.

Species of Molluscs reared on site: **Mussels**

1. General:

☐ Suspend Long Line ☐ Other Specify:

2. If you Collect Seed:

Proportion of lease presently used for spat/seed 0 %

Quantity of seed collected on lease in 2004 0 Lbs.

How many seed collectors on lease: 0

How many lines on lease: 0

3. Source of seed:

Area Purchased from

Collected by leaseholder(where)

Relayed from:

Size (mm): 0 To 0

If seed was purchased off island or in a restricted area

Introduction and Transfers Permit #:

4. Seed placed on site:

Approximate quantity of seed placed on lease in 2004 ☐ Lbs ☐ Kgs

Socking density (seed/ft): 0

Approximate distance between socks on a long line: 0 Ins.

How many socks hung on lease last fall: 0

What percent of lease was socked: 0

Approximate distance between long lines 0

5. Shellfish harvested from lease

Quantity of shellfish harvested from lease in 2004 0 Lbs.

6. Effort:

Approximately how long does it take to reach commerical size: 0 Years. Growth rates change from year to year

7. Challenges, Invasive Species and Fouling

(A) Challenge

☐ Predators ☐ Theft ☐ Ice ☐ Fouling ☐ Weather

☐ Other Please Specify:

(B) Did you see any of the following Invasive Specie

☐ Clubbed Tunicate ☐ Codium ☐ Green Crab ☐ Other Please Specify:

(C) What is your major fouling challenge:

☐ Duck Management Please Specify:

☐ Starfish Control Please Specify:

☐ Clubbed Tunicate Please Specify:

8. Suggestions and Comments.

Figure 4.2. Annual lease report for PEI mussel leaseholders.

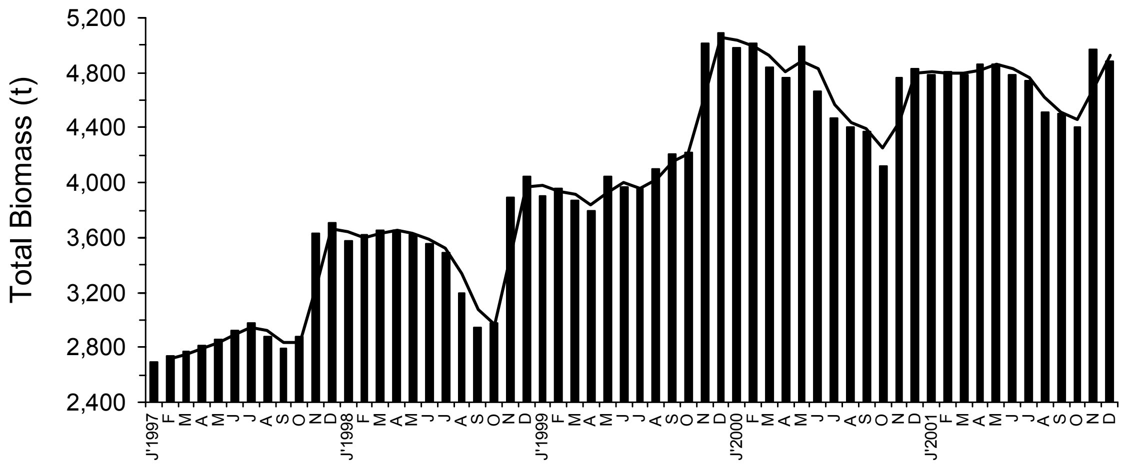


Figure 4.3. Total biomass of cultivated mussels (*Mytilus edulis*) in Tracadie Bay, PEI. Monthly values back-calculated from seed deployment and harvesting information provided by the leaseholders (Comeau et al., in preparation).

With respect to methodology, the voluntary disclosure of husbandry information is one approach recently applied as part of short-term research projects in PEI (for mussels) and NB (for oysters). The level of response was highly variable, however, and seemingly dependent upon the nature of the project. Based on this outcome, it seems that a mandatory monitoring strategy would be required in order to collect the data on a long term basis. The approach taken by DFO Charlottetown is noteworthy. In PEI, all leaseholders must complete an annual lease report as part of a license renewable process. Initially (1990s), the mail in questionnaire was meant to determine whether or not leases were utilized (active); more recently (2003), however, questions were revised to collect more detailed and standardized husbandry data (Fig. 4.2). The compliance rate in 2004, which represents the first year the revised questionnaires were sent out to leaseholders, was low (e.g. 47% for Tracadie Bay), but the DFO Charlottetown office is currently working towards increasing that number.

An alternate and perhaps complementary approach for assessing the total biomass is to perform field measurements. A sonar-GIS technique was recently used by DFO Charlottetown for compliance monitoring, i.e. to monitor the 500 sock per acre limit in Tracadie Bay. In 2005, the sonar technique was compared with simultaneous measurements made by divers. Results show that sonar can easily distinguish longlines holding mussel socks from those that have been harvested. Investigations are ongoing to determine whether sonar imagery can be used to differentiate between year-classes.

Aerial surveys represent another means for assessing shellfish inventories, at least where the culture gear is visible from the surface at some point during the production cycle. In NB, for instance, the oyster industry uses mainly floating Vexar bags, floating cages, or tables. In 2005, all oyster leases along the eastern coastline were photographed from an aircraft flying at approximately 1,000 ft. The outcome of the assessment is shown in Table 4.1. This monitoring initiative is similar to the one undertaken every five years along the coast of Normandy (Kopp et al. 2001).

Table 4.1. Number of oyster culture units in New Brunswick bays in August 2005. (L.A. Comeau, unpublished preliminary data)

	NUMBER OF CULTURE UNITS		
BAY	Floating Cages	Floating Bags	Tables
Baie St-Simon Nord		12,361	
Baie St-Simon Sud	12	42,298	
Bouctouche Bay	1,258	6,098	
Chiasson Office		8,775	
Cocagne Harbour	757	8,190	604
Lockhart Lake	40	595	
Miscou Harbour			
Néguac		20,178	
Petit Lamèque		3,370	
Pigeon Hill		6,480	
Richibucto Harbour	400	8,830	
Shediac Island	102	600	
St-Charles River	102	11,691	
Tabusintac		5,373	
Tracadie		9,838	
Village Bay	2,352	14,124	
TOTAL	5,022	158,800	604

4.2. *Shellfish performance*

Dense shellfish stocks can significantly reduce seston concentrations (see Section 3.1), possibly to levels that diminish the fitness status of natural secondary producers (Cloern, 2005). But as outlined in Section 3.1.2, the connection between shellfish aquaculture and natural secondary producers is not well understood, and consequently it is not possible at this stage to specify definite operational thresholds for particle depletion. Two potential alternative approaches were suggested in Section 3.1.2: "setting an allowable zone of effect (e.g. no measurable depletion outside the lease footprint) or allowable biological exposure time (e.g. zooplankton exposure to depleted zone limited to proportion of average life-span)." Regardless of the approach that will ultimately be selected, it requires some type of indicator for detecting particle depletion. Here, in addition those listed in Section 3.1, we consider for a moment the value of shellfish performance as a possible indicator of particle depletion. The premise is that any significant and persistent reduction in the availability of food particles will negatively impact shellfish performance. Such a condition may have recently occurred in Tracadie Bay, PEI, where leaseholders provided sufficiently detailed information to track the outcome of five seed cohorts (1995–1999). Figure 4.4 shows that many mussels from the 1998-1999 cohorts were un-harvested following the typical 24-month grow out period. In percentage terms, approximately 31% of the 1999 cohort was un-harvested compared to 7% for the 1994 and 1995 cohorts (percentage values calculated as ([biomass at 24 month / biomass at 12 month] × 100). Furthermore, a falling trend was detected in the weight of harvested socks: it gradually declined from a mean value of 9.4 kg (S.E. = 0.6) for the 1994 cohort to 6.7 kg (S.E. = 0.3) for the 1999 cohort. Together these observations point to a curtailment of growth rates and an extended production cycle, presumably due to the overstocking of shellfish and increased competition for available food resources.

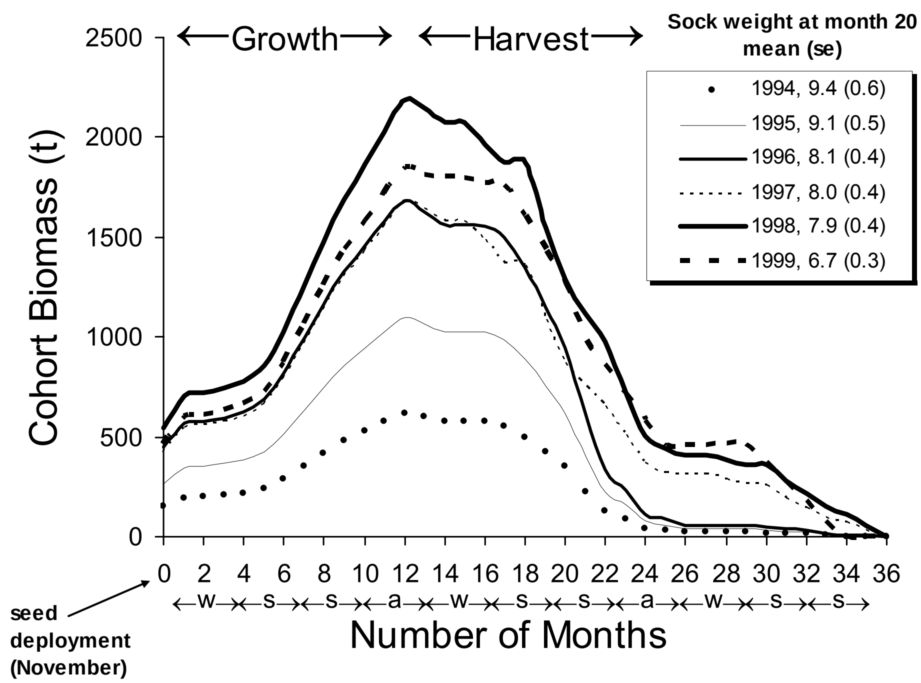


Figure 4.4. Seed deployment, growth, and harvest (Kg/sock) of several mussel cohorts (1994 to 1999) in Tracadie Bay, PEI. Trends were computed using data provided by leaseholders. Seasons identified as winter (w), spring (s), summer (s), and autumn (a) (Comeau, Davidson and Landry, unpublished data)

Although shellfish performance is certainly useful for detecting downward trends indicative of particle depletion, it may have less value for gauging the effect on natural secondary producers. An overstocking of farmed molluscs may well out-compete zooplankton populations for available food resources, perhaps to a degree where farmed molluscs will show no persistent curtailment in performance until the system is well into a HADD designation in terms of impact on natural secondary producers. An obvious drawback in such a case is that a harmful alteration to the pelagic component would go unnoticed for some time. More research is required to determine the suitability of shellfish performance for signaling the onset of a HADD.

Shellfish performance is highly variable within bays (Comeau, unpublished data), consistent with the high spatial variability in food abundance. As such, the monitoring of shellfish performance is comparable in some way to the moored instrument approach.

Both are useful for assessing the abundance of food particles over prolonged periods, but their outcome is limited to a given geographical point. However, the performance approach is less costly, meaning that the number of monitoring sites could be substantially higher compared to a moored instrument approach. In that respect, shellfish performance monitoring has bay-scale applicability; once a year, field technicians could measure shell growth in random samples taken from each lease. Plotting performance against time over many years should allow the detection of any bay-scale trends (positive or negative) if present. This approach implies that a monitoring program be set in the early phases of aquaculture development in order to document "low inventory" data points and the "natural" shellfish performance variability in the system. In some areas, shellfish aquaculture is already developed beyond the "low inventory" status. The only available references in such cases are sites located outside the leases' footprints. Because there are no cultivated molluscs beyond a lease's boundaries, special monitoring gear must be deployed both at reference and lease sites. There are standard approaches for monitoring performance in sentinel molluscs (see the Shellfish Monitoring Network, https://www.glf.dfo-mpo.gc.ca/sci-sci/smn-rmm/index-e.jsp and Réseau mollusques des rendements aquacoles, http://www.ifremer.fr/remora/). Depending on lease size, approximately eight monitoring sites (4 within lease, 4 reference sites) would be required to infer particle depletion within a lease. In summary, the shellfish performance approach has bay-scale applicability assuming that a long term monitoring program is initiated in the early phases of aquaculture development; otherwise, its applicability is downgraded to the lease scale.

It remains to be determined which specific performance indicator would best fit into a management framework. Biochemical (carbohydrates, lipids, and enzymes) indicators tend to fall into the high-cost/low-benefit category, given that they may only reflect short term nutritional status. Condition indices, commonly referred to as meat yield indices, have the disadvantage of being associated with the reproductive cycle. As a result, their use would require frequent monitoring (weekly) before, during and after spawning events in order to compute some sort of yearly standardized index. The same rationale applies to absolute weight indicators, although perhaps to a lesser extent than condition indices. The mussel sock weight, for instance, will decline during a spawning event, to the point that a submerged longline will become more noticeably buoyant. Shell length, on the other hand, is easily measured and provides growth history, which in turn has been consistently shown to correlate with food availability (Gosling 2003).

In conclusion, there is no standard program in place in Canada to monitor either shellfish farming activities or shellfish performance. Nonetheless, a number of science-based projects initiated over the past 5 years have begun gathering such data. These initiatives provide a standpoint for assessing the practicability and usefulness of certain indicators. It is recommended that a standardized monitoring of the industry inventory be conducted in parallel with any monitoring of impact indicators. In regards to shellfish performance as an indirect approach for monitoring particle depletion, its relationship to the fitness of other secondary producers must still be clarified. Further research is required to determine the suitability of shellfish performance for signaling the onset of a HADD.

5.0 CUMULATIVE EFFECTS INDICATORS AND THRESHOLDS

(P. Cranford)

Assessments of the near- and far-field environmental effects of shellfish aquaculture must consider the complexity of natural and human actions in estuarine and coastal systems. Environmental responses to multiple stressors (contaminants, fishing activities, invasive species, nutrient enrichment, sewage, climate change, coastal construction, diseases and parasites, etc.) are often intimately connected (Cleorn, 2001). Assessments of aquaculture-related habitat impacts need to account for the potentially synergistic and/or antagonistic effects on ecosystem structure and function of all anthropogenic activities. Determining the cumulative impact of multiple stressors on fish habitat requires an objective and holistic approach capable of balancing a number of potentially positive and negative environmental effects.

The NAP Working Paper entitled "*Quantification of cumulative and far-field fish habitat effects of shellfish aquaculture*" (Paper #4; Appendix I) outlines the evolution of our scientific understanding of cumulative environmental interactions leading to a contemporary conceptual model of cumulative habitat responses to shellfish aquaculture and other stressors. We continue that exercise by discussing potential habitat indicators and performance thresholds potentially capable of assessing the state of fish habitat. This is a challenge as we are often constrained by a limited understanding of how multiple stressors interact. The selection of habitat indicators and operational thresholds may therefore be best approached by focusing on general habitat attributes that are fundamental to natural ecosystem structure and dynamics. These include:

- physical processes that control the transport, mixing and flushing properties of coastal regions;
- underwater optical properties that control light availability to phytoplankton and macrophytes (primary producers); and
- chemical and biological processes that maintain the natural balance between the production and metabolism of organic matter.

The first habitat attribute, water exchange, is linked to the physical characteristics of coastal basins including tidal flushing, horizontal transport processes and bathymetry. These physical parameters determine the sensitivity of the system to single and multiple stressors and determine the inherent ability of the environment to absorb stress without the expression of a HADD. These physical processes are not readily altered by aquaculture activities, and alterations to hydrodynamic patterns have only been observed under intensive husbandry practices not presently used in Canada (e.g. Strøhmeier et al., 2005). The second attribute, light availability, is often a limiting factor controlling the production of new biomass in many coastal regions, and therefore the productivity of fish habitat. This physical property may be altered by many anthropogenic processes, including aquaculture (see Section 3.1). The third attribute encompasses a wide variety of biological and chemical processes that together control nutrient cycling and energy flow within the ecosystem. Perturbations affecting any of these processes can result in excessive food limitation (Section 3), and/or the accumulation of organic matter (Section 2) that can alter habitat productivity.

Many anthropogenic stressors can, under some conditions, change the natural balance between the production and/or metabolism of organic matter in the coastal zone. In the case of shellfish aquaculture, both can occur simultaneously as a result of shellfish grazing activity and the enhanced exchange between the water column and seabed (benthic/pelagic coupling). Shellfish aquaculture effects on these processes also interact closely with the nutrient enrichment effects of agriculture, resulting in the debate on whether or not shellfish reduce or magnify eutrophication trends in the coastal zone (see Working Paper #4). Assessing and separating the cumulative effects of aquaculture and agriculture is of particular importance for many of the extensively leased shellfish aquaculture embayments in Prince Edward Island.

The above sections have identified and discussed a number of indicators that would be suitable for assessing cumulative effects on light availability and "net organic matter balance". The former consists of a range of options from simple manual measures of light penetration depth (Secchi depth) to underwater and remote sensor measurements. The balance between the production of oxygen by microalgae (P) and oxygen respiration by bacteria (R) in the water column is a fundamental indicator of ecosystem energy flow (Section 3.4.3) and the measurement of the P/R ratio in aquaculture inlets would provide an indication of ecosystem trophic status (balance between the production and metabolism of organic matter). A potentially more practical indicator of ecosystem trophic status may be the relative abundance of bacteria and phytoplankton (B/CHL ratio) in water samples (see Section 3.4; Harrison et al., 2005). This ratio is constrained to known limits under natural coastal and oceanic conditions, but relatively high bacteria numbers have been observed in regions with extensive shellfish aquaculture, owing to depletion of phytoplankton and bacteria grazers by shellfish feeding.

Sediments generally provide a more stable integrated index of the near-field environmental changes associated with anthropogenic impacts than do water column measurements. The typical response to increased organic matter supply to the sediment is an increase in benthic oxygen demand until such time as the oxygen supply is exceeded and sub-oxic or anoxic conditions develop. Hyper-eutrophication impacts on the benthos are well documented from aquaculture and land-use practices and the cumulative effects of these two activities are relatively well known (reviewed in Working Paper #4).

The seabed is an important sink for organic matter, either from shellfish biodeposits or from other natural and anthropogenic influences (e.g. agriculture, sewage, fish processing), and plays a critical role in maintaining the natural balance between the production and metabolism of organic matter in the coastal zone. Within the oxygenated surface sediment layer, aerobic bacteria oxidize deposited organic nitrogen compounds to nitrate and nitrite (process called nitrification). This microbial degradation of deposited organic matter recycles nitrogen back to the water column, which is critical for maintaining phytoplankton stocks (Fig. 1.1). The products of nitrification (nitrate and nitrite) also diffuse into the deeper, anoxic, sediment layer where they are used for the anaerobic microbial decomposition of organic matter. The nitrogen gas that is produced by this denitrification process is not available to phytoplankton and is lost to the atmosphere (Fig. 1.1). Organic enrichment of sediments with shellfish biodeposits increases the burial of organic matter and increases the removal of nitrogen from the system by denitrification (Newell et al., 2004). This production of nitrogen gas may increase nitrogen limitation within the system, which would affect habitat productivity.

Most of the extensively leased shellfish aquaculture embayments in PEI exhibit enhanced sediment organic enrichment from both shellfish aquaculture and the eutrophication effects of nutrient enrichment from land-use (see Section 2.1 and Working Paper #4). In this case, the removal of excess nitrogen from agricultural run-off via enhanced anaerobic denitrification (caused by the deposition of organic biodeposits from shellfish; see Fig. 1.1) can be considered a positive effect. However, under conditions of excessive organic enrichment and relatively poor tidal flushing, the benthic oxygen demand can become greater than the supply, resulting in anoxic seabed conditions. This describes an important reference point, beyond which the balance between organic matter supply and nutrient recycling becomes impaired (Fig. 5.1) and the capacity of the region to assimilate further organic loading has been exceeded. First, the recycling of nitrate and nitrite back to the water column for uptake by phytoplankton is inhibited by the lack of oxygen for nitrification. Second, without a continuous supply of nitrate and nitrite from aerobic nitrification, anaerobic denitrification is also inhibited and excess nitrogen is no longer removed by this process (Figs. 1.1 and 5.1). The loss of a thin oxygenated surface sediment layer therefore has important consequences for coastal ecosystem structure and function. Reference threshold values may be defined for indicators of organic loading impacts that exceed the target value (Fig. 5.1). These could include indicator values corresponding with natural rates of nitrification and denitrification (Limit A; Fig. 5.1) and an unacceptable limit where coupled nitrification/denitrification processes are fully impaired (Limit B; Fig. 5.1).

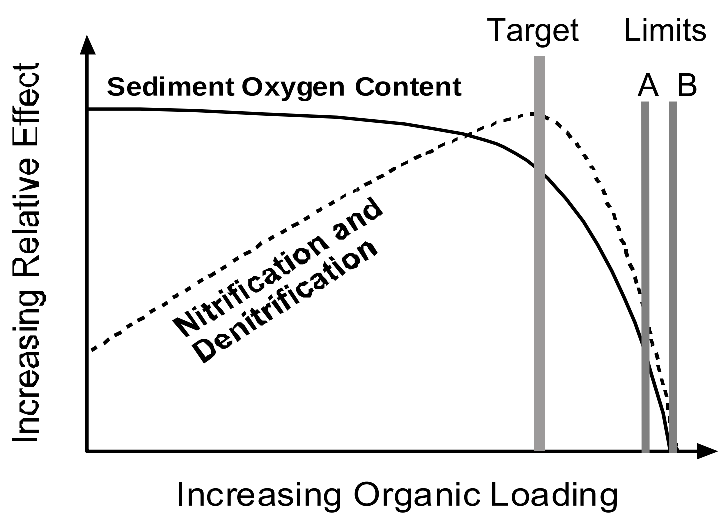


Figure 5.1. Conceptual diagram of the relative effects of organic loading (shellfish biodeposits, eutrophication, sewage, etc.) on sediment oxygen content and rates of microbial nitrification and denitrification (see Fig. 1.1). The “target” and “limit” lines are described in the text. Adapted from Newell (2004).

The determination of target and limit thresholds based on nitrification and denitrification rate measurements would be very difficult. However, the upper threshold (Limit B; unacceptable) could be defined using existing sediment geochemical criteria for anoxia and observations of the presence of white sulphur bacteria (*Beggiatoa* spp.) mats. Both approaches for monitoring sediment organic enrichment impacts are described in Section 2. Presently, biogeochemical indicators are used to assess near-field impacts close to the source(s) of the organic enrichment. To monitor for cumulative effects, a more comprehensive sampling strategy would be required that is directed at detecting the potential effects of two or more stressors impacting a larger scale (e.g. bay wide). This sampling strategy should be designed to attempt to separate indicator effects caused by each of the organic enrichment sources (e.g. to permit geospatial statistical analysis of the distribution of impacts). Intensive benthic sampling in Tracadie Bay, for example, was able to provide some insight into the relative zones of benthic impact caused by agricultural eutrophication and mussel aquaculture (see Working Paper #4).

Cumulative effects sampling programs, including a mix of benthic and water column indicators, would need to be tailored to a specific inlet and would require the direct participation of researchers to design the work required and to interpret the results. Cumulative effects monitoring programs are therefore beyond the scope of what could be practically accomplished within an aquaculture management framework and remain a topic requiring further research.

6.0 RECOMMENDED MONITORING FRAMEWORK

The Terms of Reference for this paper included the need to consider how regional and operational differences impact the applicability of tools and approaches for assessing shellfish aquaculture effects on fish habitat (Appendix 1). An important objective of this paper is therefore to provide a recommended framework of methodologies and approaches for assessing shellfish aquaculture impacts that incorporates sufficient flexibility to be of use over a wide range of culture species, husbandry practices, and environmental settings, and that is applicable to shellfish aquaculture operations that may range from less than 0.5 hectares to approximately 500 hectares. Given the highly diverse nature of the shellfish aquaculture industry in Canada, it is not sufficient to simply provide a toolbox of potential indicators and thresholds; it is equally important to make recommendations, based on sound science, as to which tools are most appropriate under different conditions.

A primary recommendation of this report is that habitat assessments could be based on a tiered approach structured on the principle that *increased environmental risk requires an increase in monitoring effort*. Various levels of monitoring could be triggered based on:

- the nature of the operation (e.g. species, culture method and stocking density per area or volume);
- the perceived environmental risk (e.g. EIA and model-based predictions);

- the ongoing measurement of environmental indicators towards verification of operational thresholds; and
- other environmental sensitivity indices (e.g. habitat sensitivity designations).

The adoption of a risk-based and responsive environmental monitoring approach is encouraged. Inherent within the recommended framework is the principle that ongoing monitoring programs be continually adaptive to changes in our state-of-knowledge concerning potential environmental impacts and indicators, and related methodological approaches. Although a comprehensive list of potential indicators of shellfish aquaculture impacts was reviewed in this paper, our discussion was weighted towards research conducted by the authors in Canada. The ICES Working Group on Marine Shellfish Aquaculture currently includes plans to combine our efforts with those of other countries. It is therefore important to maintain an ability to add or remove indicators to monitoring programs based on the evolution of our state-of-knowledge.

The recommended multi-tiered impact assessment approach described below focuses primarily on benthic marine habitat in the immediate vicinity of each shellfish aquaculture lease, thus paralleling current finfish aquaculture monitoring approaches in Canada. Scientifically defensible thresholds are available for benthic indicators and these can be used to define the hypotheses that need to be addressed in an operational monitoring program. Effective measures are also available for mitigating benthic organic enrichment impacts, and these can be linked to operational thresholds incorporated in a responsive management framework.

The DFO National Advisory Process on finfish aquaculture completed in February of 2005, showed that the aquaculture industry could not be regulated solely on the basis of site-specific observations. Local benthic geochemical and community parameters, while useful for site-specific environmental monitoring, are of limited value as indicators of changes at the ecosystem level. The pelagic and ecosystem level effects of dense shellfish populations (see Section 3 and Working Paper #4) are even more complex than for finfish culture and cannot be predicted using benthic indicator observations at single sites. Some combination of modelling and measurement of selected far-field indicators related to underwater light properties, benthic and pelagic communities, suspended particle depletion, and perhaps shellfish performance, is needed over relatively large (inlet-scale) areas to adequately assess the effects of shellfish aquaculture on fish habitat and the ecosystem. Information on the number and sizes of shellfish leases and stocking information for all farms within the management area are believed to be essential for coastal ecosystem-based assessments of shellfish aquaculture impacts on fish habitat.

The inability to adequately define quantitative operational thresholds for many valid and highly relevant indicators of habitat and ecosystem status (particularly those describing the structure and dynamics of the water column), owing to present gaps in our knowledge of ecosystems, should not preclude their potential use. Surveillance sampling programs based on selected water column parameters and shellfish performance indicators are recommended under conditions where environmental impact assessments and ongoing monitoring data indicate a relatively high risk of bay-scale impacts. Of particular concern are potential impacts on suspended particle concentrations and distribution and the resulting alterations in pelagic microflora and fauna communities and the pelagic food web. Surveillance monitoring of a suite of ecosystem traits that are thought to affect productivity, community structure and habitat (i.e. contextual indicators; Gavaris et al., 2005), is highly warranted when and where significant particle depletion by shellfish aquaculture is predicted. Seston depletion modelling capabilities have rapidly progressed in recent years and include some relatively simple quantitative assessment approaches and decision support systems (see Working Paper #2). Surveillance of pelagic indicators would compliment benthic operational monitoring and would support the basic monitoring principle of delineating cause-effect relationships.

The following subsections critically review the potential applicability of the various habitat status indicators identified in the preceding sections and outline our recommended tiered monitoring framework. This framework is specifically tailored to shellfish aquaculture and attempts to account for observed regional and species-specific differences in the environmental risks and impacts. The different monitoring tiers progress from the use of low-cost, semi-quantitative indicators, to more intensive monitoring and surveillance programs. The scientific rationale for selecting the appropriate level of monitoring is also provided.

6.1. Habitat indicator summary and assessment

An attempt was made to critically assess and summarize the applicability of a wide range of habitat indicators and measurement approaches for assessing shellfish aquaculture effects on fish habitat and productivity, based on the criteria described in Section 3.1 (e.g. regulatory requirements, species, scale, cost, benefit and responsiveness). Sections 2 to 5 of this document provide the scientific rationale for including each indicator in this assessment (i.e. address the known and potential effects of shellfish aquaculture). The ranking of indicators against specific criteria defined by the operational, practical and science-based needs of shellfish aquaculture monitoring programs is somewhat subjective in that it required professional judgements that can be weighted towards each scientist’s particular field of interest. However, efforts were made to promote impartiality and objectivity by providing opportunities for multidisciplinary input from all the authors in a consensus-building exercise.

The following tables include rankings for each of the identified habitat indicators. The capacity to address the scale of an impact was rated by indicating the maximum spatial (local, lease, or bay) scale of impact that can be detected using the identified indicator and sampling approach. A “high” ranking was reserved for indicators that provide information relevant to long-term coastal ecosystem scales. This requires indicator measurements taken at specific places and times to reflect habitat status over much larger scales. Practical considerations regarding the use of a specific indicator or measurement approach were based primarily on cost issues, specifically initial (capital and setup) and sequential (subsequent survey) costs. The latter included sampling design considerations (e.g. number of survey stations and sample size) that are reflected in the cost. The relative ranking from low to high cost is arbitrary but considers such factors as the need and availability of specialized equipment (instruments) and technical expertise. The responsiveness of the indicator/approach is a function of the time required to analyze samples and data and to interpret indicator results for managers. This was ranked from "high" (less than two weeks) to "low" (greater than three months). The latter may be considered unacceptable for a responsive management framework.

The potential benefits derived from selecting a particular indicator were assessed on a relative scale from low to high, based on a composite of sub-criteria including:

- the ability of the indicator or the specific measurement approach to detect a known habitat impact over the actual temporal and spatial scale of the effect;
- the accuracy and precision of the measured indicator value;
- the ability to specifically identify aquaculture impacts in systems exposed to multiple stressors;
- the ease of data interpretation by managers;
- the availability of established and/or defensible theoretical reference points; and
- the severity of the potential impact to ecosystem dynamics and fish habitat.

Benthic habitat indicators

Sediments provide a stable integrated index of habitat/environmental changes associated with organic enrichment from shellfish biodeposits. The different categories of benthic habitat indicators described in this paper (Section 2) are:

(1) sediment geochemical status (Eh and sulfide measurements);
(2) sediment appearance (underwater photography/video);
(3) community structure; and
(4) sediment characteristics (acoustic classification).

The progressive development of anoxic conditions with increasing organic loading from shellfish biodeposits can be detected with sediment geochemical indicator measurements (Section 2.1). These measurements can be linked to factors that control benthic community structure and ecosystem function, but no studies have looked directly at the link with secondary production (except for the ongoing project by McKindsey et al., Table 1.1). The geochemical approach is cost-effective, and changes in S and Eh around shellfish aquaculture sites are consistent with spatial and temporal observations around finfish aquaculture sites. Quantitative habitat threshold values are also available for these indicators. The only identified limitation is that S and Eh, while useful for site-specific environmental monitoring, only provide limited information on fish habitat alterations at the bay scale or at the ecosystem level. This is why we ranked this recommended approach as "medium to high" in terms of benefit, rather than giving it the highest ranking.

Several benthic population and community parameters are sensitive indicators of impacts from increasing organic loading and are therefore included as defensible indices for monitoring programs where site assessments indicate a risk of benthic impacts from shellfish biodeposits. Similarity indices (benthic community structure) and biotic indices (indicator species and trophic group distribution) are reliable and sensitive indicators of the influence of shellfish farms on benthic communities, even at low organic enrichment (see Section 2.2). The increased abundance of opportunistic deposit feeders and scavenger species and the decreased abundance of large, deep-burrowing molluscs are both effective indicators of an organic enrichment impact. However, caution is advised when interpreting results of diversity indices and biomass indicators and thus the benefit to managers of using these indicators is therefore rated as low (Table 6.1). Overall, benthic community indicators require specialized facilities, a fairly high level of expertise and considerable time for sampling, analysis and interpretation. As a result, they rank high on our cost scales and low on the responsiveness scale. Some notable exceptions include measures of the abundance of large indicators species, and the relative abundance of major phyla, which can provide early results at low cost. The results from all community indicators, as with the sediment geochemistry measures, are only applicable at the geographic scale of individual farms and little information is provided on far-field and ecosystem-level effects. These collective considerations suggest a relatively low to medium-high benefit ranking, depending on the species or community indicator in question (Table 6.1).

Benthic imaging indicators include sediment type, colour, presence of epifauna and biological structures, and the presence of bacterial mats and proportional coverage. The qualitative nature of many of these observations limits their capabilities for operational monitoring, which must address quantitative habitat impact thresholds. However, criteria are being developed for semi-quantitative image analysis that will continue to strengthen the rationale for incorporating benthic imaging in monitoring programs. Benthic imaging is a cost-effective approach for the rapid screening of benthic habitat, and the appearance of bacterial mats on the surface provides a useful and ecologically meaningful threshold indicating a transition from oxic to anoxic surface sediments. This threshold is closely linked to excessive organic enrichment from shellfish biodeposits and the associated impacts on benthic community structure (Section 2) and function (Section 5). The benefit ranking for indicators related to benthic imaging was set at medium owing to the site-specific nature of the indicators and the qualitative nature of the information (Table 6.1).

Table 6.1. Indicators of benthic habitat effects with rankings of their applicability for assessing aquaculture effects on fish habitat and productivity. Abbreviations: BCA = benthic community analysis; H' = Shannon-Weiner diversity; J = evenness; d = species richness; and IBI = index of biotic integrity. See text for details.

Indicators		**Scale**	**Cost Initial/Survey**	**Benefit**	**Responsive**
Geochemical (S & Eh)		lease	low/low	med-high	med-high
Imaging		lease	med/low	med	med-high
BCA	Diversity (# species, H', J, d)	lease	med/high	low	low
	Biotic indices (indicator species, trophic indices, IBI)	lease	med/med	med-high	med-high
	Similarity indices	lease	med/high	med	low
Acoustic classifications		bay	med/low	low	med

The overview of acoustic methods for classifying and mapping benthic habitat (Section 2.4) showed limited potential for identifying specific acoustic habitat indicators. Seabed parameters that can influence an acoustic signal are numerous, complex and variable. Although an acoustic seabed classification system tested in an aquaculture embayment in PEI was able to generate accurate and repeatable measurements of the acoustic physical characteristics of spatial variations in the seabed (i.e. effectively able to monitor bay-scale variations), there was no obvious relationship between the type of acoustic data obtained and the location of mussel culture leases. Given the present state of knowledge, we gave a "low" benefit ranking for the applicability of acoustic indicators (classifications) for use in addressing the requirements of shellfish aquaculture monitoring programs (Table 6.1). However, these technologies are developing rapidly and will likely have greater applicability to monitoring programs in the near future.

Discussions in this document of the sensitivity of the different indicators of organic enrichment impacts on benthic organisms (see Section 2.2) are highly relevant to the selection of appropriate indicators. It is important to note that, for the purposes of this document, the discussion of the relative sensitivity of indicators needs to be directed primarily at the ability of the indicator to detect the identified threshold condition(s). If decisions need to be made based on a "no change threshold", the benthic community indicators identified above are the only indicators shown to be able to detect subtle changes. However, the geochemical indicators have been shown to be suitably sensitive for detecting oxic-hypoxic-anoxic thresholds that represent important transitions in benthic community structure.

Pelagic habitat indicators

Water transport, mixing and flushing processes tend to cause considerable natural variability that can mask the pelagic impacts of shellfish aquaculture. However, shellfish aquaculture, under some conditions (largely related to hydrodynamics and shellfish stocking density), has been shown to alter many biological and chemical properties of the water column that control ecosystem structure and function (see Section 3 and Working Paper #4). Owing to the movement of the water, these impacts can be transported far-field and can alter fish habitat at the coastal ecosystem scale.

The suitability of different indicators and approaches for monitoring suspended particle depletion by shellfish aquaculture are ranked in Table 6.2. All the indicators listed are suitable for measuring particle depletion, with the qualification that total particulate matter (TPM) measurements tend to have relatively high within-sample variability, resulting in a need for greater sample replication to improve precision. Standard methodologies for chlorophyll and TPM measurements (direct analysis and sensor estimation) are readily available. The various sampling approaches each have strengths and weaknesses that will affect their applicability to monitoring programs. Manual water sampling approaches have the lowest cost, but also the lowest benefit since they are generally ineffective for detecting even lease-scale depletion. Moored instrument approaches are recommended for detecting depletion within leases, but not for bay-wide assessments. The latter requires the use of costlier technologies (e.g. CASI and BIO-

Acrobat) that depend on specialized training. The extensive data analysis and interpretation required results in some delays in delivering the results from monitoring programs (see Section 3.1). Given their usefulness for monitoring pelagic habitat changes at the bay-scale, the benefit of these approaches is highly ranked (Table 6.2). The one major caveat is that threshold values currently do not exist for particle depletion, thus reducing the usefulness of these indices for decision making.

Table 6.2. Bulk indicators of pelagic particle depletion with rankings of the applicability of available indicator measurement approaches for assessing aquaculture effects on fish habitat and productivity. TPM is total particulate matter. See text for details.

Indicator	**Approach (all indicators)**	**Scale**	**Cost Initial/Survey**	**Benefit**	**Responsive**
Chlorophyll TPM Light penetration (Secci depth) Light attenuation	Manual water sampling	local	low/low	low	high
	Moored sensors	lease	med/med	low-med	high
	Towed sensors	bay	high/med	med-high	med
	Aerial remote sensing	bay	med/med	med-high	med

There are many potential fish habitat and ecosystem consequences related to particle depletion (see Section 3 and Working Paper #4). One consequence of size-selective particle depletion by cultured shellfish is a significant change in the size structure of the microbial plankton community from larger phytoplankton to smaller picoplankton. A greater abundance of bacteria can also occur due to consumption by shellfish of some fraction of the natural grazer community (see Section 3.4). Given the potential ecosystem consequences of such a major shift in the pelagic foodweb, indicators of size spectrum changes (e.g. increased picoplankton abundance and proportion of phytoplankton; increased bacteria counts) are perceived as being highly beneficial for use in monitoring programs in extensively leased shellfish aquaculture inlets (Table 6.3). This recommendation is also related to the relatively low cost of analysis, the ease of data interpretation, and the fact that site-specific measurements of plankton community alterations generally reflect conditions over much larger scales of impact. Pelagic energy flow indicators (primary production [P] and respiration [R]) may also be useful as early warning tools for detecting a shift in the balance between the production and metabolism of organic matter. While there is a clear benefit to measuring the P/R ratio as an indicator of aquaculture impacts on pelagic energy flow, the practical utility of this indicator is reduced by the methodological expenses and the expertise necessary to interpret the results (Table 6.3), as well as the availability of an alternate indicator (bacteria/chlorophyll ratio) that provides similar information.

The benefit of monitoring harmful algal bloom (HAB) indicators, within the context of this paper, is classified as “low” in Table 6.3, owing to the present lack of evidence linking shellfish aquaculture to HABs (Section 3.5). Although there is ample evidence to link shellfish aquaculture to coastal nutrient dynamics, nutrient monitoring is speculative in nature owing to the high natural variability in the measurements (Section 3.3). Other

Table 6.3. Pelagic indicators of habitat effects with rankings of their applicability for assessing aquaculture effects on fish habitat and productivity. See text for details.

Indicator	Scale	Cost Initial/Survey	Benefit	Responsive
Bacteria counts (B)	bay	low/low	med	med
Picoplankton abundance (Pi)	bay	low/low	high	high
B/Chlorophyll ratio	bay	low/low	med	med
Pi/Chlorophyll ratio	bay	low/low	high	high
Primary production (P)	bay	high/med	med	med
Microbial respiration (R)	bay	med/med	med	med
P/R ratio	bay	high/med	med	med
Harmful algal blooms	bay	med/med	low	med
Nutrient concentrations	bay	med/low	low	high
Nutrient ratios	bay	med/low	low	high
Dissolved oxygen	bay	low/low	low	high
Shellfish performance (condition, shell length)	lease	low/low	med	high
Shellfish inventory (e.g. mean yield per sock)	site-bay	low/low	high	high

indicators of ecosystem structure and function (e.g. phytoplankton abundance and productivity and shellfish growth) may act as suitable proxies for detecting impacts on nutrient dynamics. The benefits of nutrient monitoring are therefore ranked as “low”. A similar conclusion can be drawn regarding the applicability of water column dissolved oxygen (DO) measurements as an indicator (Section 3.2). Although DO measurements provide information that is relevant to a wide range of aquaculture/ecosystem interactions, the high spatial and temporal variability limits the suitability of DO as a practical indicator of habitat status. Other indicators, such as surface sediment Eh and S (Section 2.1), can provide information related to shellfish impacts on benthic oxygen demand.

Shellfish performance indicators (Section 4), similar to bulk particle depletion measurements, do not reveal information on specific changes in the structure and functioning of ecosystems, but provide an indication as to whether shellfish aquaculture is affecting the system to a greater extent than can be absorbed by natural processes. Particle depletion and shellfish performance measurements are highly complementary, as the former provides information on food supplies that likely control the latter. A major strength is that standardized shellfish performance measures are relatively inexpensive to perform. However, the large spatial and temporal variability in particulate food supplies in coastal inlets limits the scale of impact that is represented by the performance of caged shellfish (Section 4.2). This variability in particulate food supplies was identified as the likely cause of large spatial variations in the growth rate of mussels held in cages in one PEI bay (Waite et al., 2005). Although the caged bivalve approach has potential for monitoring lease-scale effects, the interpretation of the results requires complementary information on a wide range of variables that can affect bivalve growth (temperature, currents, food abundance and nutritional quality, salinity, etc.), thereby increasing the cost of this monitoring approach.

Industry inventory and harvest production data have proven useful as indicators of growth conditions within extensively leased mussel aquaculture inlets (Section 4.1). Although there are some problems with the standardization of inventory indicator measurements and the need to account for natural annual variations in growth conditions, the cumulative data from all shellfish leases is an important resource for management, particularly if the farms occupy a large fraction of the total embayment volume. Long-term trends in total shellfish production (e.g. average mussel sock yield) have been used to assess the effects of increasing stocking density on bay-wide aquaculture production. These data are routinely collected by industry (i.e. low cost if also used in monitoring) and their benefit to addressing fish habitat issues is believed to be high, both for their value in facilitating the interpretation other indicator results, and as a general indicator for assessing bay-scale habitat effects (Table 6.3). However, industry data confidentiality has made it very difficult to employ these important data for scientific or environmental management purposes.

6.2 Recommended tiered monitoring approach

For shellfish aquaculture leases that have been assessed as having a relatively low risk to impact fish habitat, only a minimal level of monitoring appears to be warranted. The following **Level 1** monitoring program is intended to be a rapid screening method for periodic evaluations of shellfish aquaculture lease impacts. Our recommendations for Level 1 monitoring include the collection of benthic video and Secchi disk (light penetration) measurements, either annually or semi-annually at lease and suitable reference sites (Table 6.4). The choice of reference sites for comparison with lease data would include consideration of general bathymetric, hydrographic and seabed type conditions in both areas. Prior to implementing a Level 1 program at a site, it is recommended that baseline data be collected to ensure that benthic conditions are classified as Oxic (Table 2.2).

Benthic video provides a qualitative or semi-quantitative assessment of organic enrichment impacts from shellfish biodeposits, while surveillance monitoring of Secchi depth may provide insight into long-term temporal trends in water clarity related to stocking density and shellfish feeding. The primary operational threshold that could be addressed in Level 1 monitoring is the appearance of white mats of bacteria on the seabed, which are indicative of a high degree of organic enrichment and a potentially Hypoxic/Anoxic classification (Table 2.2). A change in sediment type or a colour change from tan/brown to black during the monitoring program would provide early warning of increasing organic enrichment impacts that may be used by regulators to recommend additional monitoring to more accurately document the impact or to recommend mitigations. For small, mature, low-risk shellfish leases, where Level 1 monitoring has shown no habitat changes, a decision by regulators to cease or reduce the frequency of monitoring may be scientifically warranted.

Table 6.4. Summary of habitat impact indicators recommended for use in the three monitoring tiers described in the text.

Indicators	**Monitoring Tiers**		
	Level 1	**Level 2**	**Level 3**
Benthic habitat			
Video	✓	✓	✓
S		✓	✓
Eh		✓	✓
Organic content		✓	✓
Benthic community			✓
Pelagic habitat			
Secci depth	✓	✓	✓
Chlorophyll depletion			✓
Bi:Chlorophyll			✓
Pi:Chlorophyll			✓
Stocking density/biomass		✓	✓
Mean yield per culture unit (e.g. mussel sock)			✓
Caged shellfish growth			✓

A second monitoring tier (**Level 2**) is recommended to provide annual sediment geochemistry data in cases where there are indications (predictions) or measurements of organically enriched seabed conditions known to be deleterious to fish habitat. Specific indicator recommendations for Level 2 monitoring programs are as follows. Bottom video and Secchi disk depths are recommended as described for the Level 1 program. In addition, the collection of sediment cores or grabs is recommended. The need to conduct statistical analysis of data requires a minimum of three sediment grab/core samples to be collected at a minimum of three sampling stations within the lease area and at a reference site located at a suitable area adjacent to the lease (Table 6.4). Surface (2-cm depth) sediment should be analyzed for redox, total sulfides, organic content and water content. Sampling would be best conducted annually in late summer/early fall when the biological oxygen demand of surface sediments is greatest. An annual shellfish inventory report for the lease would greatly benefit this Level 2 program. Under the recommended responsive management framework, regulators would have the ability to increase or decrease the level of monitoring required for any site during subsequent sampling cycles, based, at least in part, on the review of Level 2 program results.

The third recommended monitoring tier (**Level 3**) was designed, and is recommended, for assessing sites that are predicted to present a relatively high environmental risk and/or the results of ongoing Level 2 monitoring at the site show degrading habitat status (benthic performance thresholds exceeded). Specific recommendations for the Level 3 monitoring program are as follows. In addition to conducting all components of the Level 2 program (video, Secchi disk depth, geochemistry and bivalve inventory), the annual collection of other habitat data is recommended, including:

- benthic community analysis;
- magnitude and spatial extent of pelagic particle depletion (e.g. chlorophyll);
- pelagic community structure (e.g. picoplankton/CHL and bacteria/CHL ratios); and
- characterization of bivalve performance indices within each lease (Table 6.4).

Sampling may also need to be increased to give greater spatial detail, depending on the monitoring plan developed by the regulator, on a site-specific basis. As with the other monitoring tiers, it is recommended that regulators have the capacity to alter the level of monitoring required for any site during subsequent sampling cycles, based on the review of the Level 3 program results.

The Environmental Management Plan (EMP) for the construction, operation, and decommissioning of suspended mussel aquaculture facilities in St. Ann's Harbour, Cape Breton, NS (Stuart, 2003), is an example of the current "state of the art" in shellfish aquaculture monitoring in Canada. This is the largest shellfish aquaculture operation in Canada and the EMP was designed to reflect the risks identified in the Environmental Impact Assessment, some of which were predicted using hydrodynamic and particle depletion models. This program contains some similarities to the Level 3 monitoring program outlined above. The annual monitoring program, conducted each September or October, consists of the following components:

1) Documentation of the aquaculture activities and characteristics at each lease:
    - Number and time of year each year class is introduced
    - Number of mussel long lines
    - Number of units (socks) per longline
    - Average number of mussels per unit
    - Estimated total biomass
    - Degree of biofouling
    - Degree of product falloff (if any)
    - Estimated mortality
    - Incidence of predation
    - Incidences of biotoxins, diseases and pathogens
2) Documentation of mussel performance information at each lease:
    - Average length and weight of mussels within each year class
    - Mussel condition indices for each year class
    - Average mussel yield per sock
    - Total harvest weight of mussels sold from each year class
3) Documentation of benthic performance information within and adjacent to each lease:
    - Sediment organic matter content
    - Sediment water content
    - Sediment total sulfides
    - Sediment Eh potential
    - Underwater video survey

4) Documentation of water column environment information within and adjacent to each lease:
    - Weekly Secchi disk measurements
    - Incidences of toxic algal blooms
    - Incidences of migratory birds
    - Biophysical sensor profiles, including CTD, turbidity and chlorophyll
    - Measures of ammonia concentration
    - Vertical measures of turbidity

The St. Ann's Harbour EMP is based on adaptive and responsive management principles, such that the type, level and frequency of subsequent monitoring may vary depending on the results generated from prior monitoring activities and advances in scientific research. This EMP employed the environmental quality objectives (operational thresholds) recommended by the Aquaculture Association of Nova Scotia (2002). Predefined adjustments to industry activities (remediation responses) are specified in the EMP based on sediment sulfide thresholds. Data from other indicators are used along with the sulfide data to help support operational and habitat management decisions. These decisions are assisted by a combined data management and decision-support tool that was developed by DFO Science with input from DFO Habitat Management and the Nova Scotia Department of Agriculture and Fisheries (NSDAF). The tool is based on GIS (ArcView) mapping of all collected environmental monitoring data. All reference and lease sampling sites are classified by the software relative to oxic-hypoxic-anoxic benthic performance thresholds, allowing managers to quickly determine if there is a need to recommend any predefined remediation measures (Fig. 6.1).

The current high level of aquaculture monitoring in St. Ann's Harbour was the direct result of the potential risks associated with this large project. This EMP is reviewed here to illustrate that some of our recommendations related to high-risk sites are already in practice in Canada. An example of the implementation of a lower level of monitoring (approximately equivalent to Level 2) is the recent survey by the NSDAF of aquaculture leases in Nova Scotia.

6.3. Decision thresholds and responsive management

Threshold values for various benthic indicators have been proposed and are recommended to describe changes in benthic habitats in response to organic enrichment (Table 6.5). Unlike any other monitoring approach currently available, the use of biogeochemical measures for assessing habitat status (i.e. performance, quality) is well advanced for aquaculture impact studies. The results are immediately available to managers and farm operators and provide information on the oxic status of benthic conditions at their site and indicate whether there is a need for remediation (i.e. responsive management). Redox (Eh) and sulfide (S) threshold levels have been incorporated into the Environmental Management Guidelines for salmon aquaculture in New Brunswick (NBDFA, 2000), and provide regulators with a fair, accurate, and objective method of ensuring compliance with regulatory requirements. As shown in data summarized from various locations (Table 2.2), the same procedures are being widely used in provincial EMP programs associated with finfish and shellfish aquaculture site monitoring. This approach has been advocated by the Office of Sustainable Aquaculture

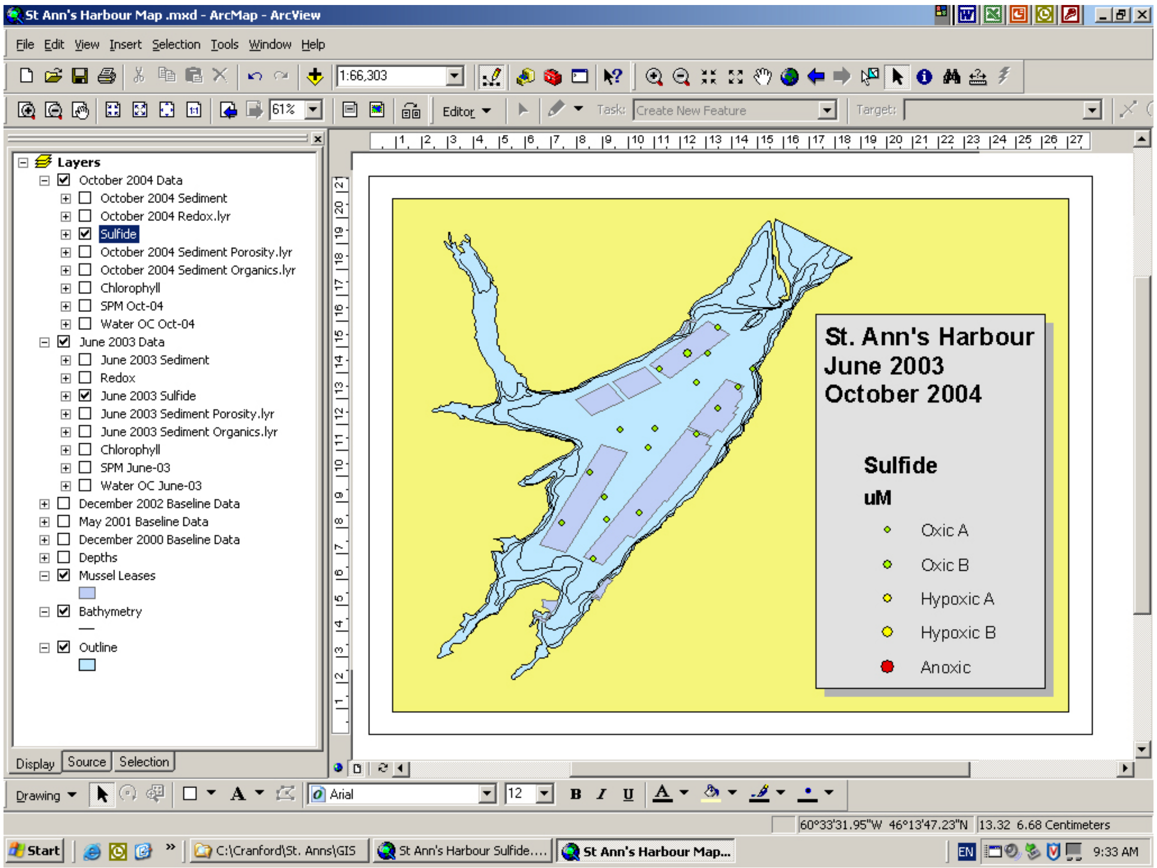


Figure 6.1. Average benthic sulfide concentrations in St. Ann's Harbour viewed using the ArcView GIS data base and decision-support tool. The size and colour of the circle plotted at each sampling site is proportional to sulfide concentration. Threshold sulfide levels are used to classify benthic conditions and as regulatory triggers.

(DFO, 2002) and is currently used by Habitat Management in the Maritimes Region as criterion for site evaluation.

Threshold values for sulfides and Eh potentials are useful since they provide operational targets for monitoring programs and reference points for decision making. The recommended target is for S and Eh levels to remain within the natural range as indicated by measurements at the reference site(s). Wildish *et al.* (2001) have recently formalized ranges of benthic habitat variables for use in quantifying oxic-anoxic conditions in sediments and these have been re-examined in Section 2.1 (Tables 2.2 and 2.3). Based on results summarized in Table 2.3, a new threshold S level separating Hypoxic A and Hypoxic B sediments of 3000 μM is recommended for use within a responsive shellfish aquaculture management framework to encourage the site operator to prevent site conditions from reaching an anoxic state (Table 6.5). The use of this limit would allow a site operator an opportunity to implement mitigation measures before sediments are allowed to become anoxic. Any mitigation actions would be best implemented as soon as possible and options include husbandry practice alterations, changes to site configuration and operational changes (e.g. reduce stocking biomass). Threshold values identified in Table 6.5 (defined by transitions between organic enrichment groups) would serve as valuable benchmarks for supporting decisions on site management and the required level of monitoring needed in subsequent habitat assessment surveys to adequately document habitat status.

Table 6.5. Comparison of characteristic effects of organic enrichment in normal (oxic) to highly organically rich (anoxic) sediments. The application of SPI (Sediment Profile Imagery) and BHQ (Benthic Habitat Quality Index) to define organic enrichment gradients in marine sediments is described in Wildish et al. (2004a). Terminology and limits of ranges of S concentrations defining five organic enrichment groups (in bold) are modified from Wildish et al. (2001, 2004a) based on maximum median S concentrations and Eh potentials using data in Table 2.2 and summarized in Table 2.3.

Type of Measure	**Organic Enrichment Group**				
Microbial	Normal	Oxic	Hypoxic	Anoxic	
Macrofauna	Normal	Transitory	Polluted	Grossly Polluted	
SPI (BHQ)	>10	5-10	2-4	<2	
Geochemical	***Oxic A***	***Oxic B***	***Hypoxic A***	***Hypoxic B***	***Anoxic***
Eh (mV_{NHE})	**>+100**	**+100 to -50**	**-50 to -100**	**-100 to -150**	**<-150**
$S^{=}$ (uM)*	**<750**	**750 to 1500**	**1500 to 3000**	**3000 to 6000**	**>6000**

*mean values of lower and upper $S^{=}$ concentrations representing ranges in K-means clustered data based on observations from 13 locations presented in Table 2.3.

7.0 REFERENCES

AANS, 2002. Design of the environmental monitoring program for the marine aquaculture industry in Nova Scotia. Prepared for the Aquaculture Association of Nova Scotia by CoastalSmith Inc.

Anderson, D.M., D.M. Kulis, J.J. Sullivan, S. Hall, and C. Lee. 1990. Dynamics and physiology of saxitoxin production by the dinoflagellates *Alexandrium* spp. Mar. Biol. 104: 511-524.

Anderson, M.R., M.F. Tlusty, and V.A. Pepper. 2005. Organic enrichment at cold water aquaculture sites - the case of coastal Newfoundland. Environmental effects of marine finfish aquaculture, Handbook of environmental chemistry, Vol. 5, Springer, Berlin, p. 99-113.

Archambault, P., K. Banwell, and A.J. Underwood. 2001. Temporal variation in the structure of intertidal assemblages following the removal of sewage. Mar. Ecol. Prog. Ser. 222: 51-62

Barranguet, C., E. Alliot, and M.-R. Plante-Cuny. 1994. Benthic microphytic activity at two Mediterranean shellfish cultivation sites with reference to benthic fluxes. Oceanol. Acta 17: 211-221.

Bacher, C., J. Grant, A.J.S. Hawkins, J. Fang, M. Zhu, and M. Besnard. 2003. Modelling the effect of food depletion on scallop growth in Sungo Bay (China). Aquat. Living Resour. 16: 10-24.

Bates, S.S. 2004. Amnesic shellfish poisoning: domoic acid production by *Pseudo-nitzschia* diatoms. Aqua Info Aquaculture Notes 16, 4 p.

Bates, S.S. 1997. Toxic phytoplankton on the Canadian east coast: implications for aquaculture. Bull. Aquacult. Assoc. Canada 97-3: 9-18.

Bates, S.S., J. Worms, and J.C. Smith. 1993. Effects of ammonium and nitrate on domoic acid production by *Pseudonitzschia pungens* in batch culture. Can. J. Fish. Aquat. Sci. 50: 1248-1254.

Bates, S.S., D.L. Garrison, and R.A. Horner. 1998. Bloom dynamics and physiology of domoic-acid-producing *Pseudo-nitzschia* species. In: D.M. Anderson, A.D. Cembella, and G.M. Hallegraeff (eds.). The physiological ecology of harmful algal blooms. Springer-Verlag, Heidelberg, p. 267-292.

Baudinet, D., E. Alliot, B. Berland, C. Grenz, M. Plante-Cuny, R. Plante, and C. Salen-Picard. 1990. Incidence of mussel culture on biogeochemical fluxes at the sediment-water interface. Hydrobiol. 207: 187-196.

Bec, B., J.Husseini-Ratrema, Y. Collos, P. Souchu, and A. Vaquer. 2005. Phytoplankton seasonal dynamics in a Mediterranean coastal lagoon: emphasis on the picoeukaryotic community. J. Plank. Res. 27: 881-894.

Boyd, A.J. and K.G. Heasman. 1998. Shellfish mariculture in the Benguela system: water flow patterns within a mussel farm in Saldanha Bay, South Africa. J. Shellfish Res. 17: 25-32.

Brooks, K.M. and C.V.M. Mahnken. 2003. Interactions of Atlantic salmon in the Pacific northwest environment II. Organic wastes. Fish. Res. 62: 255-293.

Brooks, K.M., A.R. Stierns, C.V.W. Mahnken, and D.B. Blackburn. 2003. Chemical and biological remediation of the benthos near Atlantic salmon farms. Aquaculture 219: 355-377.

Brown, J.R., R.J. Gowen, and D.S. McLusky. 1987. The effect of salmon farming on the benthos of a Scottish sea loch. J. Exp. Mar. Biol. Ecol. 109: 39-51.

Bugden, J.B.C., B.T. Hargrave, P.M. Strain, and A.R.J. Stewart. 2001. Spatial patterns of some physical and chemical variables in Passamaquoddy Bay and Letang Inlet, southwestern Bay of Fundy, September 1999. Can. Tech. Rep. Fish. Aquat. Sci. 2356: iv + 96 p.

Butler, D.J. 2003. Possible impacts of marine farming of mussels (*Perna canaliculus*) on king shags. DOC Science Internal Series 111, Department of Conservation, Wellington, New Zealand, 29 p. http://www.doc.govt.nz/Publications/004~Science-and-Research/DOC-Science-Internal-Series/PDF/DSIS111.pdf

Calabrese, E.J. and L.A. Baldwin. 1999. Reevaluation of the fundamental dose-response relationship. BioScience 49: 725-732.

Callier, M.D., C.W. McKindsey, and G. Desrosiers. 2005. Influence de la biodéposition de moules sur l'environnement benthique de deux sites mytilicoles: Havre-aux-Maisons et Grande-Entrée Réseau d'Aquaculture du Québec (RAQ), September 12, Quebec (QC), Canada.

Caswell, H. and J.E. Cohen. 1991. Communities in a patchy environment: a model of disturbance, competition, and heterogeneity. In: S.T.A. Kjap (ed.). Ecological heterogeneity. Vol 86, Springer-Verlag, New York, p. 97-122.

Cembella, A.D. and E. Todd. 1993. Seafood toxins of algal origin and their control in Canada. In: I.R. Falconer (ed.). Algal toxins in seafood and drinking water. Academic Press, London, p. 129-144.

Chamberlain, J., T.T. Fernandes, P. Read, T.D. Nickell, and I.M. Davis. 2001. Impacts of biodeposits from suspended mussel (*Mytilus edulis* L.) culture on the surrounding surficial sediments. J. Mar. Sci. 58: 411-416.

Christensen P.B., R.N. Glud, T. Dalsgaard, and P. Gillespie. 2003. Impacts of longline mussel farming on oxygen and nitrogen dynamics and biological communities of coastal sediments. Aquaculture 218: 567-588.

Cloern, J.E. 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquatic Ecology 33: 1-16.

Cloern, J.E. 2005. Ecological consequences of food competition between bivalves and zooplankton. In: 8th International Conference on Shellfish Restoration. Oct. 2-5, 2005, Brest, France. p. 71 (abstract).
http://www.ifremer.fr/icsr05/en/doc/abstract-book-150905.pdf

Cohen J. 1988. Statistical power analysis for the behavioural sciences, 2nd ed. Hills-Dale.

Collins, W.T. and K.P. Rhynas. 1998. Acoustic seabed classification using echo sounders: operational considerations and strategies. Proc. Can. Hydrogr Conf. Victoria, Canada: 384-390.

Courties, C., A. Vaquer, J. Lautier, M. Troussellier, M.J. Chrétiennot-Dinet, J. Neveux, C. Machado, and H. Claustre. 1994. Smallest eukaryotic organism. Nature 370: 255.

Cranford, P., M. Dowd, J. Grant, B. Hargrave, and S. McGladdery. 2003. Ecosystem level effects of marine bivalve aquaculture, p. 51-95. In: Fisheries and Oceans Canada. A scientific review of the potential environmental effects of aquaculture in aquatic systems. Volume 1. Far-field environmental effects of marine finfish aquaculture (B.T. Hargrave); Ecosystem levels effects of marine bivalve aquaculture

(P. Cranford, M. Dowd, J. Grant, B. Hargrave, and S. McGladdery); Chemical use in marine finfish aquaculture in Canada: a review of current practices and possible environmental effects (L.E. Burridge). Can. Tech. Rep. Fish. Aquat. Sci. 2450: ix + 131 p.

Cranford, P.J. and K. Lee. 2005. A perspective on offshore oil and gas environmental effects monitoring, p. 1-21. In: Armsworthy, S.L., P.J. Cranford, and K.Lee (eds.). Offshore oil and gas environmental effects monitoring. Battelle Press, Columbus.

Crawford, C.M., C.K.A. Macleod, and I.M. Mitchell, 2003. Effects of shellfish farming on the benthic environment. Aquaculture 224: 117-140.

Cullen P. (1990) The turbulent boundary between water science and water management. Freshwater Biol. 24: 201-209.

D'Amours O. and P. Archambault. 2005. Monitoring fishes and macroinvertebrates to determine indirect influence of bivalve aquaculture on ecosystem productivity. AquaNet V, Victoria, BC.

Dahlback, B. and L.A.H. Gunnarsson. 1981. Sedimentation and sulfate reduction under a mussel culture. Mar. Biol. 63: 269-275.

Dame, R.F. 1996. Ecology of marine bivalves: an ecosystem approach. CRC Press, Boca Raton, Forida.

Dame, R., N. Dankers, T. Prins, H. Jongsma, and A. Smaal. 1991. The influence of mussel beds on nutrients in the Western Wadden Sea and Eastern Scheldt estuaries. Estuaries 14: 130-136.

Danovaro R., C. Gambi, G.M. Luna, and S. Mirto. 2004. Sustainable impact of mussel farming in the Adriatic Sea (Mediterranean Sea): evidence from biochemical, microbial and meiofaunal indicators. Mar. Poll. Bull. 49: 325-333.

De Casabianca, M.-L., T. Laugier, and D. Collart. 1997. Impact of shellfish farming eutrophication on benthic macrophyte communities in the Thau Lagoon, France. Aquacult. Int. 5: 301-314.

DFO, 2002. Aquaculture site application guide: review process and guidelines. Office of Sustainable Aquaculture, 18 January, 2002.

DFO, 2003. Salmon holding capacity in southwestern New Brunswick. Can. Tech. Rep. Fish. Aquat. Sci. 2489: iv + 60 p.

Diaz, R.J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanogr. Mar. Biol. Ann. Rev. 33: 245-303.

Diaz, R.J., M. Solan, and R.M. Valente. 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. J. Environ. Man. 73: 165-181.

Dowd, M., F.H. Page, R. Losier, P. McCurdy, and G. Burgden. 2001. Physical oceanography of Tracadie Bay, PEI: analysis of sea level, current, wind and drifter data. Can. Tech. Rep. Fish. Aquat. Sci. 2347:iii + 77 p.

Dowd, M. 2003. Seston dynamics in a tidal inlet with shellfish aquaculture: a model study using tracer equations. Est. Coast. Shelf Sci. 57:523-537.

Dowd, M. 2005. A bio-physical coastal ecosystem model for assessing environmental effects of marine bivalve aquaculture. Ecol. Model. 183: 323-346.

Downes, B.J., L.A. Barmuta, P.G. Fairweather, D.P. Faith, M.J. Keough, P.S. Lake, B.D. Mapstone, and G.P. Quinn (eds.). 2002. Monitoring ecological impacts: concepts and practice in flowing waters. Cambridge University Press, Cambridge.

Drouin, A., P. Sirois, and P. Archambault. 2006. Structure des communautés d’invertébrés et des espèces d'amphibiens dans des lacs avec et sans omble de fontaine (*Salvelinus fontinalis*) en forêt boréale. Rapp. tech. can. sci. halieut. aquat. 2628 vii:40.

Ducklow, H.W. 2001. Bacterioplankton. In: Steele, J.H., S.A. Thorpe, and K.K. Turekian (eds.). Encyclopaedia of ocean sciences. Academic Press, San Diego, p. 217-224.

Duplisea, D.E., S. Jennings, K.J. Warr, and T.A. Dinmore. 2002. A size-based model of the impacts of bottom trawling on benthic community structure. Can. J. Fish. Aquat. Sci. 59: 1785-1795.

Dupuy, C., A. Vaquer, T. Lam-Hoai, C. Rougier, N. Mazouni, J. Lautier, Y. Collos, and S. Le Gall. 2000. Feeding rate of the oyster *Crassostrea gigas* in a natural planktonic community in the Mediterranean Thau Lagoon. Mar. Ecol. Prog. Ser. 205:171-184.

Edgar, G.J., C.K. MacLeod, R.B. Mawbey, and D. Shields. 2005. Broad-scale effects of marine salmonid aquaculture on macrobenthos and the sediment environment in southeastern Tasmania. J. Exp. Mar. Biol. Ecol. 327:70-90.

Fairweather, P.G. 1999. Determining the ‘health’ of estuaries: priorities for ecological research. Aust. J. Ecol. 24: 441-451.

Fraschetti, S., C.N. Bianchi, A. Terlizzi, G. Fanelli, C. Morri, and F. Boer. 2001. Spatial variability and human disturbance in shallow subtidal hard substrate assemblages: a regional approach. Mar. Ecol. Prog. Ser. 212:1-12.

Fréchette, M., D.A. Booth, B. Myrand, and H. Bermard. 1991. Variability and transport of organic seston near a mussel aquaculture site. ICES Mar. Sci. Symp. 192: 24-32.

Fuentes, J., Gregoria, V., Giraldez, R., and Molares, J. 2000. Within-raft variability of the growth rate of mussels, *Mytilis galloprovincialis*, cultivated in the Ría de Arousa (Spain). Aquaculture 189: 39-52.

Gavaris, S., J.M. Porter, R.L. Stephenson, G. Robert, and D.S. Pezzack. 2005. Review of management plan conservation strategies for Canadian fisheries on Georges Bank: a test of a practical ecosystem-based framework. ICES CM 2005/BB:05.

Gavine, F. M. and McKinnon, L. J. 2002. Environmental Monitoring of Marine Aquaculture in Victorian Coastal Waters: A Review of Appropriate Methods. Technical Report No. 46. Marine and Freshwater Resources Institute, Victoria, Australia.

GESAMP (Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). 1991. *Global Strategies for Marine Environmental Protection.* Reports and Studies, No. 45, 34 p.

GESAMP (Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). 1996. Monitoring the ecological effects of coastal aquaculture. Reports and Studies, No. 57, 38 p.

Gillespie G., R. Mackereth, D. Greig, D. Ming, M. Boivin, and T. Allison. 2002. Design standards for improving fish habitat management. Can. Manuscript Rep. Fish. Aquat. Sci. 2592: vi + 89 p.

Glasby, T.M. 1997. Analysing data from post-impact studies using asymetrical analyses of variance: a case study of epibiota on marinas. Aust. J. Ecol. 22: 448-459.

Gosling, E. 2003. Bivalve molluscs: biology, ecology and culture. Blackwell Publishing, Oxford.

Grant, J., A, Hatcher, D.B. Scott, P. Pocklington, C.T. Schafer, and G.V. Winters. 1995. A multidisciplinary approach to evaluating impacts of shellfish aquaculture on benthic communities. Estuaries 18: 124-144.

Grant, J., P. Cranford, B. Hargrave, M. Carreau, B. Schofield, S. Armsworthy, V. Burdett-Coutts, and D. Ibarra 2005. A model of aquaculture biodeposition for multiple estuaries and field validation at blue mussel (*Mytilus edulis*) culture sites in eastern Canada. Can. J. Fish. Aquat. Sci. 62: 1271-1285.

Grant, J., C. Bacher, P. Cranford, T. Guyondet, and M. Carreau. 2006. A spatially explicit ecosystem model of seston depletion in dense mussel culture. In preparation.

Gray, J.S. 1981. The ecology of marine sediments: an introduction to the structure and function of benthic communities. Cambridge Studies in Modern Biology 2: 185 p.

Gray, J.S., D. Calamari, R. Duce, J.E. Portmann, P.G. Wells, and H.L. Windom. 1991. Scientifically based strategies for marine environmental protection and management. Mar. Poll. Bull. 22: 432-440.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologist. New York, John Wiley.

Haigh, R. and F.J.R. Taylor. 1990. Distribution of potentially harmful phytoplankton species in the northern Strait of Georgia, British Columbia. Can. J. Fish. Aquat. Sci. 47: 2339-2350.

Hargrave, B.T. (ed.). 2002. Environmental studies for sustainable aquaculture (ESSA): 2002 Workshop Report. Can. Tech. Rep. Fish. Aquat. Sci. 2411: v + 117 p.

Hargrave, B.T. 2003. Far-field environmental effects of marine finfish aquaculture, p. 1-49. In: Fisheries and Oceans Canada. A scientific review of the potential environmental effects of aquaculture in aquatic systems. Volume 1. Far-field environmental effects of marine finfish aquaculture (B.T. Hargrave); Ecosystem levels effects of marine bivalve aquaculture (P. Cranford, M. Dowd, J. Grant, B. Hargrave and S. McGladdery); Chemical use in marine finfish aquaculture in Canada: a review of current practices and possible environmental effects (L.E. Burridge). Can. Tech. Rep. Fish. Aquat. Sci. 2450: ix + 131 p.

Hargrave, B.T. 1994. A benthic enrichment index, p. 79-91. In: B.T. Hargrave (ed.). Modelling benthic impacts of organic enrichment from marine aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

Hargrave, B.T. 2002. Thresholds for assessing organic matter assimilative capacity in marine sediments, p. 22-28. In: B.T. Hargrave (ed.). Environmental studies for sustainable aquaculture (ESSA): 2002 workshop report. Can. Tech. Rep. Fish. Aquat. Sci. 2411: v + 117 p.

Hargrave, B.T., D.E. Duplesia, E. Pfeiffer, and D.J. Wildish. 1993. Seasonal changes in benthic fluxes of dissolved oxygen and ammonium associated wit marine culture Atlantic salmon. Mar. Ecol. Prog. Ser. 96: 249-257.

Hargrave, B.T., G.A. Phillips, L.I. Doucette, M.J. White, T.G. Milligan, D.J. Wildish, and R.E. Cranston. 1997. Assessing benthic impacts of organic enrichment from marine aquaculture. Water Air Soil Pollut. 99: 641-650.

Harrison W.G., T. Perry, and W.K.W. Li. 2005. Ecosystem indicators of water quality. Part I. Plankton biomass, primary production and nutrient demand. In: B.T. Hargrave (ed.). Environmental effects of marine finfish aquaculture. Handbook of environmental chemistry, Vol. 5, Part M: 59-82.

Hartog, F. 2003. Impacts of scallop dredging on habitat structure and demersal fish community of the Magdalen Shelf, Canada. MSc. Thesis, Univ. Laval, Quebec.

Hartstein, N.D. and A.A. Rowden, 2004. Effect of biodeposits from mussel culture on macroinvertebrate assemblages at sites of different hydrodynamic regime. Mar. Environ. Res. 57: 339-357.

Hatcher, A., J. Grant, and B. Schofield. 1994. Effects of suspended mussel culture (*Mytilus* spp.) on sedimentation, benthic respiration and sediment nutrient dynamics in a coastal bay. Mar. Ecol. Prog. Ser. 115: 219-35.

Heasman, K.G.M, G.C. Pitcher, C.D. McQuaid, and T. Hecht. 1998. Shellfish mariculture in the Benguela system: raft culture of *Mytilus galloprovincialis* and the effect of rope spacing on food extraction, growth rate, production, and condition of mussels. J. Shellfish Res. 17:33-39.

Holmer, M., D. Wildish and B.T. Hargrave. 2005. Organic enrichment from marine finfish aquaculture and effects on sediment processes. In: B.T. Hargrave (ed.). Environmental effects of marine finfish aquaculture, Chapter 9, Handbook of environmental chemistry, Vol. 5, Springer, Berlin, p. 181-206.

Hopkins, W.G. 2000. A new view of statistics, http://www.sportsci.org/resource/stats/index.html Accessed January 28, 2006.

Ibarra, D.A. 2003. Estimation of seston depletion by cultured mussels (*Mytilus* spp.) using measurements of diffuse attenuation of solar irradiance from optical measurements. MSc. Thesis, Dalhousie University, Halifax, NS, 127 p.

ICES. 2000. Report of the Working Group on Environmental Interactions of Mariculture. ICES CM 2000/F:02. Ref.: ACME+E. Copenhagen, Denmark. http://www.ices.dk/products/CMdocs/2000/F/F0200.PDF

ICES. 2004. Report of the Study Group to Review Ecological Quality Objectives for Eutrophication. ICES CM 2004/ACE:04. Ref. ACME, C, E. Copenhagen, Denmark. http://www.ices.dk/reports/ACE/2004/SGEUT04.pdf

ICES. 2004. Report of the Working Group on Marine Shellfish Culture (WGMASC). ICES CM 2004/F:02 Ref. ACME. http://www.ices.dk/reports/MCC/2004/WGEIM04.pdf

ICES. 2005. Report of the Working Group on Environmental Interactions of Mariculture (WGEIM). ICES CM 2005/F:04 Ref. I, ACME http://www.ices.dk/reports/mcc/2005/WGEIM05.pdf

Inglis, G.J., B.J. Hayden, and A.H. Ross. 2000. An overview of factors affecting the carrying capacity of coastal embayments for mussel culture. National Institute of Water & Atmospheric Research Ltd, Christchurch, p. 31.

Jamison, G.S, L. Chew, G. Gillespie, A. Robinson, L. Bendell-Young, W. Heath, B. Bravender, A. Tompkins, D. Nishimura, and P. Doucette. 2001. Phase 0 review of the environmental impacts of intertidal shellfish aquaculture in Baynes Sound. DFO Can. Sci. Advis. Sec. Res. Doc. 2001/125: i + 103 p. http://www.dfo-mpo.gc.ca/csas/Csas/DocREC/2001/RES2001_125e.pdf

Jaramillo, E., C. Bertran, and A. Bravo. 1992. Mussel biodeposition in an estuary in southern Chile. Mar. Ecol. 82: 85-94.

Jones, C.G., J.H. Lawton, and M. Shachak. 1994. Organisms as ecosystem engineers. Oikos 69: 373-386.

Jones G.P and U.L. Kaly. 1996. Criteria for selecting marine organisms in biomonitoring studies. In: Schmitt, RJ. and C.W. Osenberg (eds.). Detecting ecological impacts: concepts and applications in coastal habitats. Academic Press, p. 29-48.

Karlsson, O., P.R. Jonsson, and A.I. Larsson. 2003. Do large seston particles contribute to the diet of the bivalve *Cerastoderma edule*? Mar. Ecol. Prog. Ser. 261: 164-173.

Kaspar, H.F., P. Gillespie, L.F. Boyer, and A.L. MacKenzie. 1985. Effects of mussel aquaculture on the nitrogen cycle of benthic communities in Kenepuru Sound, Marlborough Sound, New Zealand. Mar. Biol. 85: 127-136.

Kautsky, N. and S. Evans. 1987. Role of biodeposition by *Mytilus edulis* in the circulation of matter and nutrients in a Baltic coastal ecosystem. Mar. Ecol. Prog. Ser. 38: 201-212.

Kenny, A. J., I. Cato, M. Desprez, G. Fader, R. T. E. Schüttenhelm, and J. Side. 2003. An overview of seabed-mapping technologies in the context of marine habitat classification. ICES J. Mar. Sci. 60: 411-418.

Keough, M.J. and B.D. Mapstone. 1997. Designing environmental monitoring for pulp mills in Australia. Water Sci. Tech. 35: 397-404.

Keough M.J. and G.P. Quinn. 1991. Causality and the choice of measurements for detecting human impacts in marine environments. Aust. J. Mar. Freshwater Res. 42: 539-554.

Kepkay P.E., W.G. Harrison, and J.B.C. Bugden. 2005. Ecosystem indicators of water quality. Part II. Oxygen production and oxygen demand. In: B.T. Hargrave (ed.) Environmental effects of marine finfish aquaculture. Handbook of environmental chemistry, Vol. 5, Part M: 83-97.

Kopp J., G. Messian, E. Le Gagneur, F. Cornette, and M. Ropert. 2001. État des stocks conchylicoles Normands en 2000. DRV/RA/RTS/2001-08 Port-en-Bessin, 50 p.

Lawrence, J.E., A.G. Bauder, M.A. Quilliam, and A.D. Cembella. 1998. *Prorocentrum lima*: a putative link to diarrhetic shellfish poisoning in Nova Scotia, Canada. In: Reguera, B., Blanco, J., Fernández, M. L., and Wyatt, T. (eds.). Harmful algae. Xunta de Galicia and the Intergovernmental Oceanographic Commission of UNESCO, p. 78-79.

Lea, L. 2002. Mussel aquaculture management plan for Tracadie Bay, Prince Edward Island. February 2002. 6 p.

Li, W.K.W. 2002. Macroecological patterns of phytoplankton in the northwestern North Atlantic Ocean. Nature 419: 154-157.

Li, W.K.W., E.J.H. Head, and W.G. Harrison. 2004. Macroecological limits of heterotrophic bacterial abundance in the ocean. Deep-Sea Research I 51: 1529-1540.

Li, W.K.W., W.G. Harrison, and E.J.H. Head. 2006. Phytoplankton monitoring in Bedford Basin, the Scotian Shelf and the Labrador Sea: a large scale multi-year coherence. Atlantic Zone Monitoring Program Bulletin 5.

Lloyd, B.D. 2003. Potential effects of mussel farming on New Zealand's marine mammals and seabirds: a discussion paper. DOC Science Internal Series 111, Department of Conservation, Wellington, New Zealand, 35 p. http://www.doc.govt.nz/Publications/004~Science-and-Research/Miscellaneous/PDF/musselfarms.pdf

Magurran, A.E. 2004. Measuring biological diversity. Blackwell Publishing.

Mapstone, B.D. 1995. Scalable decision rules for environmental impact studies: effect size, Type I and Type II errors. Ecol. Appl. 5: 401-410.

Mallet, A.L. and B. Myrand. 1995. The culture of the blue mussel in Atlantic Canada. Cold-water aquaculture in Atlantic Canada, In: Boghen, A.D. (ed.). The Canadian Institute for Research on Regional Development, Moncton, NB, p. 255-296.

Martin. J.L. and M.M. LeGresley. 1998. Does aquaculture impact harmful algal blooms in the southwest Bay of Fundy? In: M.D.B. Burt and P.G. Wells (eds.). Coastal monitoring and the Bay of Fundy. Proceedings of the Maritime Atlantic Ecozone Science Workshop, held in St. Andrews, New Brunswick, Nov. 11-15, 1997. p. 24-27.

Mattsson, J. and O. Lindén. 1983. Benthic macrofauna succession under mussels, *Mytilus edulis* L. (Bivalvia), cultured on hanging long-lines. Sarsia 68: 97-102

Mazouni N., J.-C. Gaertner, J.-M. Deslous-Paoli, S. Landrien, and M. G. d'Oedenberg 1996. Nutrient and oxygen exchanges at the water-sediment interface in a shellfish farming lagoon (Thau, France). J. Exp. Mar. Biol. Ecol. 205: 91-113.

McDonald, L.L. and W.P. Erickson. 1994. Testing for bioequivalence in field studies: has a disturbed site been adequately reclaimed? In: Fletcher, D.J. and B.F.J. Manly (eds.). Statistics in ecology and environmental monitoring. University of Otago Press, Dunedin, p. 183-197.

Meeuwig, J.J, J.B. Rasmussen, and R.H. Peters, 1998. Turbid waters and clarifying mussels: their moderation of empirical chl:nutrient relations in estuaries in Prince Edward Island, Canada. Mar. Ecol. Prog. Ser. 171: 139-50.

Mirto, S., T. La Rosa, R. Danovaro, and A. Mazzila. 2000. Microbial and meiofaunal response to intensive mussel-farm biodeposition in coastal sediments of the Western Mediterranean. Mar. Pollut. Bull. 40: 244-252.

Møhlenberg, F. and M.V. Riisgård. 1978. Efficiency of particle retention in 13 species of suspension-feeding bivalves. Ophelia 17: 239-246.

NBDAFA. 2000. Environmental management guidelines for the Atlantic salmon marine cage aquaculture industry in New Brunswick. New Brunswick Dept. of Agriculture and Aquaculture.

Navarro, E., J.I.P. Iglesias, A. Perez Camacho, U. Labarta, and R. Beiras. 1991. The physiological energetics of mussels (*Mytilus galloprovincialis* Lmk) from different cultivation rafts in the Ria de Arosa (Galicia, N.W. Spain). Aquaculture 94:, p. 197-212.

Newell, R.I.E. 2004. Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. J. Shellfish Res. 23: 51-61.

Nilsson, H.C. and R. Rosenberg. 1997. Benthic habitat quality assessment of an oxygen stressed fjord by surface and sediment profile images. J. Mar. Syst. 11: 249-264.

Nilsson, H.C. and R. Rosenberg. 2000. Succession in marine benthic habitats and fauna in response to oxygen deficiency: analyzed by sediment profile-imaging and by grab samples. Mar. Ecol. Prog. Ser. 197: 139-149.

Ogilvie, S.C., A.H. Ross, and D.R. Schiel. 2000. Phytoplankton biomass associated with mussel farms in Beatrix Bay, New Zealand. Aquaculture 181: 71-80.

Page, F.H., R. Losier, P. McCurdy, D. Greenberg, J. Chaffey, and B. Chang 2005. Dissolved oxygen and salmon cage culture in the southwestern New Brunswick portion of the Bay of Fundy. Hdb. Env. Chem. 5 (Part M): 1-28.

Pearson, T.H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev. 16: 229-311.

Pearson, T.H. and K.D. Black. 2001. The environmental impacts of marine fish cage culture. In: Black, K.D. (ed.). Environmental impacts of aquaculture, Sheffield Academic Press, p. 1-32.

Pilditch, C.A., J. Grant, and K.R. Bryan. 2001. Seston supply to sea scallops (*Placopecten magellanicus*) in suspended culture. Can. J. Fish. Aquat. Sci. 58: 241-253.

Plante-Cluny, M.-R., R. Plante, N. Mazouni, M.-F. Fontaine, P. Souchu, J.-M. Deslous-Paoli, and C. Grenz. 1998. Oxygen fluxes involving the benthic micro-and macro-phytic components in the Thau Lagoon under pre-anoxic conditions. Oceanol. Acta 21: 819-829.

Preston, J.M. and W.T. Collins. 2000. Bottom classification in very shallow water by high-speed data acquisition. Proceedings of MTS/IEEE Oceans 2000 (Sept. 11-14), Providence, USA :1277-1282. http://marine.questertangent.com/papers/bottomclass.pdf

Preston, J.M., W.T. Collins, D.C. Mosher, R.H. Poeckert, and R.H. Kuwahara. 1999. The strength of correlations between geotechnical variables and acoustic classifications. Proceedings of MTS/IEEE Oceans'99, Seattle, USA, 6 p.

Prins, T.C. and A.C. Smaal. 1990. Benthic-pelagic coupling: the release of inorganic nutrients by an intertidal bed for *Mytilus edulis*. In: M. Barnes and R.N. Gibson (eds.). Trophic relationships in the marine environment. Aberdeen University Press, Aberdeen, Scotland, p. 89-103.

Prins, T.C. and A.C. Smaal. 1994. The role of the blue mussel *Mytilus edulis* in the cycling of nutrients in the Oosterschelde estuary (The Netherlands). Hydrobiologia 282/283: 413-429.

Prins, T.C., A.C. Smaal, and R.F. Dame. 1998. A review of the feedbacks between bivalve grazing and ecosystems processes. Aquat. Ecol. 31: 349-359.

QTC Impact. 2004. Acoustic seabed classification for echo sounders: user manual and references (Version 3.4). Quester Tangent Corporation, Sidney, B.C., Canada, 147 p.

Ragnarsson, S.A. and D. Raffeelli. 1999. Effects of the mussel *Mytilus edulis* L. on the invertebrate fauna of sediments. J. Exp. Mar. Biol. Ecol. 24: 31-43.

Rice, M.A. 2001. Environmental impacts of shellfish aquaculture: filter feeding to control eutrophication. In: Tlusty, M.F., D.A. Bengston, H.O. Halvorson, S.D. Oktay, J.B. Pearce, and R.B. Rheault, Jr. (eds.). Marine aquaculture and the environment: a meeting for stakeholders in the Northeast. Cape Cod Press, Falmouth, MA, USA, p. 77-86. http://neaq2.securesites.net/scilearn/research/pdf/12__.pdf

Rosenberg, R. and L.O. Loo. 1983. Energy-flow in a *Mytilus edulis* culture in Western Sweden. Aquaculture 35: 151-161.

Rydberg, L., B. Sjoberg, and A. Stigebrandt. 2003. The interaction between fish farming and algal communities of the Scottish Waters - a review. Scottish Executive. Environment Group Research Report 2003/04, 57 p. http://www.scotland.gov.uk/Resource/Doc/46930/0014748.pdf

Sabol, B.M. and S.A. Johnston. 2001. Innovative techniques for improved hydroacoustic bottom tracking in dense aquatic vegetation. Aquatic Plant Control Research

Program, U.S. Army Corps of Engineers, Washington, DC, ERDC/EL MP-01-2, 16 p.

Sakami, T., K. Abo, K. Takayanagi, and S. Toda. 2003. Effects of water mass exchange on bacterial communities in an aquaculture area during summer. Est. Coastal Shelf Sci. 56: 111-118.

Schaanning, M.T. and P. Kupka Hansen. 2005. The suitability of electrode measurement for assessment of benthic organic impact and their use in a management system for marine fish farms. In: B.T. Hargrave (ed.). Environmental effects of marine finfish aquaculture, Handbook of environmental chemistry, Vol. 5, Springer, Berlin, p. 381-408.

Scottish Executive. 2002. Review and synthesis of the environmental impacts of aquaculture. The Scottish Association for Marine Science and Napier University. Scottish Executive Central Research Unit, 325 p. http://www.scotland.gov.uk/cru/kd01/green/reia.pdf

Shaw, K.R. 1998. PEI benthic survey. Tech. Rep. Environ. Sci. No. 4, Dept. of Environment and Dept. of Fisheries and Environment, Charlottetown, 75 p.

Shumway, S.E., C. Davis, R. Downey, R. Karney, J. Kraeuter, J. Parsons, R. Rheault, and G. Wikfors. 2003. Shellfish aquaculture – In praise of sustainable economies and environments. World Aquacult. 34: 15-19.

Sigg, L. 2000. Redox potential measurements in natural waters: significance, concepts and problems. In: Schuring, J., H.D. Schulz, W.R. Fischer, J. Bottcher, and W.H.M. Duijnivel (eds). Redox fundamentals, processes and applications. Springer, Berlin, p 1-12.

Silvert, W. 2001. Impact on habitats: determining what is acceptable. In: Tlusty, M.F., D.A. Bengston, H.O. Halvorson, S.D. Oktay, J.B. Pearce, and R.B. Rheault, Jr. (eds.). Marine aquaculture and the environment: a meeting for stakeholders in the Northeast. Cape Cod Press, Falmouth, MA, USA, p. 16-40. http://neaq2.securesites.net/scilearn/research/pdf/12__.pdf

Smaal, A.C. and T.C. Prins. 1993. The uptake of organic matter and the release of inorganic nutrients by bivalve suspension feeder beds. In: R.F. Dame (ed). Bivalve filter feeders in estuarine and coastal ecosystem processes. North Atlantic Treaty Organization ASI Series, Volume G33, Springer-Verlag, Heidelberg, p. 271-298.

Smayda, T.J. 1990. Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic. In: Granéli, E., B. Sundström, L. Edler, and D.M. Anderson (eds.). Toxic marine phytoplankton. Elsevier Science Publishing Co., Inc., New York, p. 29-40.

Smith, J.C., R. Cormier, J. Worms, C.J. Bird, M.A. Quilliam, R. Pocklington, R. Angus and L Hanic. 1990. Toxic blooms of the domoic acid containing diatom Nitzschia pungens in the Cardigan River, Prince Edward Island. In: Granéli, E., Sundström, B., Edler, L., and Anderson, D.M. (Eds.) Toxic marine phytoplankton. Elsevier Science Publishing Co., Inc., New York, pp. 227-232.

Solan, M., J.D. Germano, D.C. Rhoads, C. Smith, E. Michaud, D. Parry, F. Wenzhöfer, B. Kennedy, C. Henriques, E. Battle, D. Carey, L. Iocco, R. Valente, J. Watson, and R. Rosenberg. 2003. Towards a greater understanding of pattern, scale and process in marine benthic systems: a picture is worth a thousand worms - Review article. J. Exp. Mar. Biol. Ecol. 285-286: 313-338.

Sommer, U. 1994. Are marine diatoms favoured by high Si:N ratios*?* Mar. Ecol. Prog. Ser. 115: 309-315.

Souchu, P., A. Vaquer, Y. Collos, S. Landrein, J.-M. Deslous-Paoli, and B. Bibent. 2001. Influence of shellfish farming activities on the biogeochemical composition of the water column in Thau Lagoon. Mar. Ecol. Prog. Ser. 218: 141-152.

Staniford, D. 2003. Closing the net on sea cage fish farming. Presented at “Charting the Best Course: The Future of Mariculture in Austraila’s Marine Environment” (27th August 2003), a conference organised by the Queensland Conservation Council and the Australian Marine Conservation Society.

Stenton-Dozey, J.M.E., L.F. Jackson, and A.J. Busby. 1999. Impact of mussel culture on macrobenthic community structure in Saldanha Bay, South Africa. Mar. Poll. Bull. 39: 357-366.

Stenton-Dozey, J.M.E., T. Probyn, and A.J. Busby. 2001. Impact of mussel (*Mytilus galloprovincialis*) raft-culture on benthic macrofauna, in situ oxygen uptake, and nutrient fluxes in Saldanha Bay, South Africa. Can. J. Fish. Aquat. Sci. 58: 1021-1031.

Stigerbrandt, A. 1991. Computations of oxygen fluxes through the sea surface and the net production of organic matter with application to the Baltic and adjacent seas. Limnol. Oceanogr. 36: 444-454.

Strain, P.M. 2002. Nutrient dynamics in Ship Harbour, Nova Scotia. Atmosphere-Ocean 40: 45-58.

Strain, P.M. 2005. Eutrophication impacts of marine finfish aquaculture. DFO Can. Sci. Advis. Sec. Res. Doc. 2005/034: ii + 37 p.

Strain, P.M. and B.T. Hargrave. 2005. Salmon aquaculture, nutrient fluxes and ecosystem processes in southwestern New Brunswick. In: B.T. Hargrave (ed.). Environmental effects of marine finfish aquaculture, Handbook of environmental chemistry, Vol. 5, Springer, Berlin, p. 29-57.

Strohmeier, T., J. Aure, A. Duinker, T. Castberg, A. Svardal, and Ø. Strand. 2005. Flow reduction, seston depletion, meat content and distribution of diarrhetic shellfish toxins in a long-line blue mussel (*Mytilus edulis*) farm. J. Shellfish Res. 24: 15-23.

Stuart, R. 2003. Environmental Management Plan for the Bounty Bay Shellfish Inc. and 5M Aqua Farms mussel aquaculture operation in St. Ann’s Harbour, Cape Breton, NS.

Sutherland, T.F., C.D. Levings, S.A. Peteresen, D. Sinclair, S. Jepps, J. Knight, B. Gillard, W.Knapp, and R. McPhie. 2005. Data report on sediment properties and macrofaunal communities collected at Sir Edmund Bay, British Columbia. Can. Data Rep. Fish. Aquat. Sci. 1165: v + 21 p.

Sutherland, T.F., C.D. Levings, R. McPhie, S.A. Petersen, and W. Knapp. 2006. A benthic study examining the relationship between sediment properties and faunal groups observed at Sir Edmund Bay, British Columbia. Can. Tech. Rep. Fish. Aquat. Sci. (in press).

von Szalay, P.G. and R.A. McConnaughey. 2002. The effect of slope and vessel speed on the performance of a single beam acoustic seabed classification system. Fish. Res. 56: 99-112.

Taekema, B., K. White, and E. McGreer. 2006. Salmon aquaculture environmental data report - results of sampling program for year 2005. British Columbia Ministry of Environment, Nanaimo, BC (in prep.).

Tett, P. and V. Edwards. 2002. Review of harmful algal blooms in Scottish coastal waters. Report to the Scottish Environment Protection Agency. Aqua KE Government Documents 2004:5240060. http://sepa.org.uk/pdf/aquaculture/projects/habs/habs_report.pdf

Trites, R.W. and L. Petrie. 1995. Physical oceanographic features of Letang Inlet including evaluation and results from a numerical model. Can. Tech. Rep. Hydro. Ocean Sci. 163: iv + 53 p.

Trousseillier, M., G. Cahet, P. Lebaron, and B. Baleux. 1993. Distribution and dynamics of bacterial production in relation to wind perturbations in a Mediterranean lagoon. Limnol. Oceanogr. 38: 193-201.

Tsutsumi, H. 1990. Population persistence of *Capitella* sp. (Polychaeta; Capitellidea) on a mud flat subject to environmental disturbance by organic enrichment. Mar. Ecol. Prog. Ser. 63: 147-156.

Underwood, A.J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. Aust. J. Mar. Freshwater Res. 42: 569-587.

Underwood A.J. 1993a. How not to design an environmental monitoring program: a case study from the FAC (up in Botany Bay). Aust. Biol. 6: 194-197.

Underwood, A.J. 1993b. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Aust. J. Ecol. 18: 99-116.

Underwood A.J. 2000. Trying to detect impacts in marine habitats: comparisons with suitable reference areas. In: Sparks, T. (ed.). Statistics in ecotoxicology. John Wiley & Sons, LTD, Toronto, p. 279-308.

Underwood A.J. and M.G. Chapman. 2003. Power, precaution, Type II error and sampling design in assessment of environmental impacts. J. Exp. Mar. Biol. Ecol. 296: 49-70.

Vaquer, V., M. Trousseillier, C. Courties, and B. Bibent. 1996. Standing stock and dynamics of picophytoplankton in the Thau Lagoon (northwest Mediterranean coast). Limnol. Oceanogr. 41: 1821-1828.

Waddington, T. and K. Hart. 2003. Tools and techniques for the acquisition of estuarine benthic habitat data. Science Applications International Corporation, Report 628: 52 p. http://www.csc.noaa.gov/benthic/mapping/techniques/pdf/tooltech.pdf

Warwick, R.M. and K.R. Clarke. 1990. A statistical analysis of coral community response to the El Niño events of 1982-3 in the Thousand Islands, Indonesia. Coral Reefs 8: 171-179.

Warwick, R.M. and K.R. Clarke. 1991. A comparison of some methods for analysing changes in benthic community structure. J. Mar. Biol. Ass. U.K. 71: 225-244.

Warwick, R.M. and K.R. Clarke. 1993. Increased variability as a symptom of stress in marine communities. J. Exp. Mar. Biol. Ecol. 172: 215-226.

Washington, H.G. 1984. Diversity, biotic and similarity indices. Water Res. 18: 653-694.

Weston, D.P. 1990. Quantitative examination of macrobenthic community changes along an organic enrichment gradient. Mar. Ecol. Prog. Ser. 61: 233-244.

Wildish, D.J. and G.W. Pohle. 2005. Benthic macrofauna changes resulting from finfish mariculture. In: B.T. Hargrave (ed.). Environmental effects of marine finfish aquaculture, Handbook of environmental chemistry, Vol. 5, Springer, Berlin, p. 275-304.

Wildish, D.J., Akagi, H.M., N. Hamilton, and B.T. Hargrave. 1999. A recommended method for monitoring sediments to detect organic enrichment from mariculture in the Bay of Fundy. Can. Tech. Rep. Fish. Aquat. Sci. 2286: iii + 31 p.

Wildish, D.J., B.T. Hargrave, and G. Pohle. 2001. Cost effective monitoring of organic enrichment resulting from salmon mariculture. J. Mar. Sci. 58: 469-476.

Wildish, D.J., J.E. Hughes-Clark, G.W. Pohle, B.T. Hargrave, and L.M. Mayer. 2004a. Acoustic detection of organic enrichment in sediments at a salmon farm is confirmed by independent groundtruthing methods. Mar. Ecol. Prog. Ser. 267: 99-105.

Wildish, D.J., H.M. Akagi, B.T. Hargrave, and P.M. Strain. 2004b. Inter-laboratory calibration of redox and total sulfide measurements in interfacial marine sediments and the implications for organic enrichment assessment. Can. Tech. Rep. Fish. Aquat. Sci. 2546: iii + 25 p.

Wildish, D.J., G.W. Pohle, B.T. Hargrave, T.F. Sutherland, and M.R. Anderson. 2005. Benthic monitoring methods for habitat management of finfish aquaculture in Canada. Can. Sci. Adv. Sect. Res. Doc. 2005/039, 39 p.

Wilkinson, L. 2000. Systat 10 (for Windows). SPSS, Chicago, IL.

Yang, C.Z. and L.J. Albright. 1994. The harmful phytoplankter *Chaetoceros concavicornis* causes high mortalities and leucopenia in chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*). Can. J. Fish. Aquat. Sci. 51: 2493-2500.

Yokoyama, H. 2002. Impact of fish and pearl farming on the benthic environments in Gokasho Bay: evaluation from seasonal fluctuations of the macrobenthos. Fish. Sci. 68: 258-268.

Appendix 1

Terms of Reference (8 November, 2005)

National Peer-Review Workshop

Aquaculture-Environment Interactions: Shellfish Aquaculture in the Marine Environment

February 28 – March 3, 2006
Moncton, N.B.

Chairperson: Jake Rice

Preamble

A national workshop has been initiated to meet DFO Habitat Management's need for science advice related to the fish habitat effects of shellfish aquaculture in the marine environment.

Science advice is required to define the fish habitat effects of shellfish aquaculture, to define appropriate tools and methodologies (models, indicators and thresholds) for predicting and assessing these effects, to determine the sensitivity of selected fish habitats to these effects, and to enhance national coherence between regional decision-making approaches.

In addition to addressing farm-scale habitat effects under the *Fisheries Act*, there is also a need to consider the importance of ecosystem-scale and cumulative effects of shellfish aquaculture activities... This consideration of ecosystem effects requires sound science to inform decision-making. Developing a fully integrated ecosystem-based approach, however, is beyond the scope of this process and would detract from its focus; rather the goal here is to provide advice on shellfish aquaculture effects that will be immediately applicable to Habitat Management site-specific decision-making and that may be pertinent to forthcoming ecosystem management initiatives. An attempt will be made, where scientific knowledge exists, to consider tools and methodologies for assessing both farm-scale and ecosystem-wide effects.

This advice on tools and methodologies will assist DFO's Habitat Management in reviewing shellfish aquaculture site applications and in assessing ongoing aquaculture operations in the marine environment. In addition, it will provide a basis for any future, more detailed examinations of aquaculture activities in a site-specific, or an ecosystem-based management context.

Science Advice

Peer-reviewed science advice will be generated to, where possible, address the following questions:

A. What are the positive and the negative effects (benthic and/or water column) of marine shellfish aquaculture on fish habitat? How do shellfish aquaculture effects on fish habitat differ from the 'natural' effects of wild shellfish?
   What are the effects of the physical structures used in shellfish aquaculture on fish habitat (including lines, socks, bags, predator control devices, etc.)? How can these effects be assessed or measured?

B. What chemical, biological or physical indicators developed and in use for monitoring the farm-scale fish habitat effects of marine finfish aquaculture are applicable to monitoring shellfish aquaculture effects? Describe the thresholds that apply. What other habitat indicators are available specifically to measure these shellfish aquaculture effects? What are the thresholds for these potential indicators?
   > ***NOTE:*** *A threshold should be defined as a point where significant changes to fish habitat can be identified. It is equally important for the science advice to identify the threshold as it is to describe the change to habitat that is associated with the threshold.*

C. What modeling methodologies or techniques are available to provide predictions of the potential effects of shellfish aquaculture operations on the marine environment? What are the advantages and disadvantages of these methodologies or techniques?

D. What are the cumulative and far-field effects of shellfish aquaculture in fish habitat? How can the cumulative fish habitat effects of shellfish aquaculture (*e.g.* marine eutrophication, oxygen or phytoplankton depletion, community shifts, exceeding carrying capacity) be quantified?
   What tools or indicators are useful for quantifying the far-field or ecosystem-scale fish habitat effects of shellfish aquaculture? What are the advantages and disadvantages of these tools or indicators?

E. What types of fish habitat are likely to be affected by shellfish aquaculture? How sensitive (in relative or absolute terms) are these habitats to shellfish aquaculture effects?

Working Papers

The following working papers, each focusing on different themes, will inform the development of science advice. Where applicable, all working papers should include consideration of how regional and operational differences impact the applicability of tools and approaches for assessing shellfish aquaculture effects on fish habitat.

Paper #1 *Identification of shellfish aquaculture effects on fish habitat*

- Overview of bivalves; shellfish aquaculture; ecological role of bivalves in natural habitat
- Identification of effects of shellfish aquaculture on fish habitat

Paper #2 *Indicators and thresholds to assess the effects of shellfish aquaculture on fish habitat*

- Benthic, pelagic and shellfish performance indicators and thresholds, including near-, far-field and cumulative effects
- Monitoring frameworks for assessing fish habitat effects of shellfish aquaculture and methodologies, including case studies

Paper #3 *Modeling approaches to assess the potential effects of shellfish aquaculture on the fish habitat*

- Modeling near-field benthic effects of shellfish farms (using DEPOMOD)
- Biogeochemical modelling; ecosystem-based modelling

Paper #4 *Cumulative and far-field fish habitat effects of shellfish aquaculture*

- Identification of far-field effects (i.e. types, extent and consequences), and cumulative effects of shellfish aquaculture on fish habitat
- Determination of likelihood of far-field and cumulative effects

Paper #5 *Determination of habitat sensitivity to shellfish aquaculture effects*

- Case studies exploring the sensitivity of shellfish aquaculture on fish habitat:
    - Effects of bottom oyster aquaculture on eelgrass
    - Intertidal shellfish aquaculture in Baynes Sound
    - Seabed classification in a mussel farming bay

Process and Outputs

The latest date for submission of working papers is **February 10, 2006**. The peer-review workshop is planned for **February 28 – March 3, 2006** in Moncton, New Brunswick. Invited participants will receive copies of the working papers approximately two weeks prior to the meeting.

At the meeting a review of the papers will seek to determine whether the conclusions presented in the working papers are credible, supported by scientific data and complete relative to global knowledge. Peer-reviewed advice to address the questions posed by DFO Habitat Management will also be developed.

Within four weeks of the meeting, papers will be revised, and published advice will be provided to Habitat Management in the form of a Canadian Science Advisory Secretariat (CSAS) Science Advisory Report. Discussions and results from the workshop will also be documented in a CSAS Proceedings document.