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A Scientific Review of the Potential Environmental Effects of Aquaculture in Aquatic Ecosystems

Volume I:
Far-field Environmental Effects of Marine Finfish Aquaculture
(B.T. Hargrave);
Ecosystem Level 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).

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Chemical Use in Marine Finfish Aquaculture in Canada: A Review of Current Practices and Possible Environmental Effects
(L.E. Burridge)

EDITORIAL BOARD

FOREWORD

Context

The Government of Canada is committed to ensuring the responsible and sustainable development of the aquaculture industry in Canada. The Minister of Fisheries and Oceans' announcement of the $75 M Program for Sustainable Aquaculture (PSA), in August 2000, is a clear expression of this commitment. The objective of the PSA is to support the sustainable development of the aquaculture sector, with a focus on enhancing public confidence in the sector and on improving the industry's global competitiveness. Ensuring the sector operates under environmentally sustainable conditions is a key federal role.

As the lead federal agency for aquaculture, Fisheries and Oceans Canada (DFO) is committed to well-informed and scientifically-based decisions pertaining to the aquaculture industry. DFO has an ongoing program of scientific research to improve its knowledge of the environmental effects of aquaculture. The department is also engaged with stakeholders, provinces and the industry in coordinating research and fostering partnerships. As a contribution to the Federal government's Program for Sustainable Aquaculture, DFO is conducting a scientific review of the potential environmental effects of aquaculture in marine and freshwater ecosystems.

Goal and Scope

Known as the State-of-Knowledge (SOK) Initiative, this scientific review provides the current status of scientific knowledge and recommends future research studies. The review covers marine finfish and shellfish, and freshwater finfish aquaculture. The review focuses primarily on scientific knowledge relevant to Canada. Scientific knowledge on potential environmental effects is addressed under three main themes: impacts of wastes (including nutrient and organic matter); chemicals used by the industry (including pesticides, drugs and antifoulants); and interactions between farmed fish and wild species (including disease transfer, and genetic and ecological interactions).

This review presents potential environmental effects of aquaculture as reported in the scientific literature. The environmental effects of aquaculture activities are site-specific and are influenced by environmental conditions and production characteristics at each farm site. While the review summarizes available scientific knowledge, it does not constitute a site-specific assessment of aquaculture operations. In addition, the review does not cover the effects of the environment on aquaculture production.

The papers target a scientific and well-informed audience, particularly individuals and organizations involved in the management of research on the environmental interactions of aquaculture. The papers are aimed at supporting decision-making on research priorities, information sharing, and interacting with various organizations on research priorities and possible research partnerships.

Each paper was written by or under the direction of DFO scientists and was peer-reviewed by three experts. The peer reviewers and DFO scientists help ensure that the papers are up-to-date at the time of publication. Recommendations on cost-effective, targeted research areas will be developed after publication of the full series of SOK review papers.

State-of-Knowledge Series

DFO plans to publish 12 review papers as part of the SOK Initiative, with each paper reviewing one aspect of the environmental effects of aquaculture. This Volume contains three papers: Far-field environmental effects of marine finfish aquaculture; Ecosystem level effects of marine bivalve aquaculture; Chemical use in marine finfish aquaculture in Canada: a review of current practices and possible environmental effects.

Further Information

For further information on a paper, please contact the senior author. For further information on the SOK Initiative, please contact the following:

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AVANT-PROPOS

Contexte

Le gouvernement du Canada est déterminé à assurer le développement responsable et durable de l'industrie aquacole au Canada. Le Programme d'aquaculture durable (PAD) de 75 millions de dollars annoncé par le ministre des Pêches et des Océans en août 2000 traduit clairement cet engagement. Ce programme vise à soutenir le développement durable du secteur aquacole, surtout en améliorant la confiance du public envers l'industrie et la compétitivité globale de celle-ci. Veiller à ce que l'industrie fonctionne dans des conditions durables sur le plan environnemental constitue une responsabilité essentielle du gouvernement fédéral.

À titre d'organisme fédéral responsable de l'aquaculture, Pêches et Océans Canada (MPO) est déterminé à prendre des décisions éclairées qui reposent sur des données scientifiques éprouvées en ce qui concerne l'industrie aquacole. Le MPO mène un programme de recherches scientifiques pour améliorer ses connaissances sur les effets de l'aquaculture sur l'environnement. Le Ministère collabore également avec des intervenants, les provinces et l'industrie à la coordination des recherches et à l'établissement de partenariats. Le MPO contribue au Programme de l'aquaculture durable du gouvernement fédéral en passant en revue la littérature scientifique qui aborde les effets possibles de l'aquaculture sur les écosystèmes marins et d'eau douce.

Objectif et portée

Désignée projet sur l'état des connaissances, cette revue de la littérature définit l'état actuel des connaissances scientifiques sur les effets de l'élevage de poissons et de mollusques en mer et de la pisciculture en eau douce et fait des recommandations de recherches futures. La revue, qui se concentre surtout sur les connaissances scientifiques applicables au Canada, les aborde sous trois thèmes principaux : les impacts des déchets (éléments nutritifs et matière organique), les produits chimiques utilisés par l'industrie (pesticides, médicaments et agents antisalissures) et les interactions entre les poissons d'élevage et les espèces sauvages (transfert de maladies et interactions génétiques et écologiques).

Cette revue présente les effets environnementaux possibles de l'aquaculture documentés dans la littérature scientifique. Les effets environnementaux des activités aquacoles dépendent du site, des conditions environnementales et des caractéristiques de production de chaque établissement aquacole. L'examen résume les connaissances scientifiques disponibles mais ne constitue pas une évaluation des activités aquacoles spécifique au site. L'examen ne porte pas non plus sur les effets de l'environnement sur la production aquacole.

Les articles sont destinés à un auditoire de scientifiques et de personnes bien informées, notamment des personnes et des organisations participant à la gestion de la recherche sur les interactions environnementales de l'aquaculture. Les articles visent à soutenir la prise de décision sur les priorités de recherche, la mise en commun de l'information et les interactions entre diverses organisations concernant les priorités de recherche et les partenariats de recherche possibles.

Rédigées par des scientifiques du MPO ou sous leur supervision, les articles ont été contrôlés par des pairs, ce qui assure qu'ils sont à jour au moment de leur publication. Après la publication de toute la série d'articles sur l'état des connaissances, des recommandations de recherches ciblées et rentables seront faites.

Série sur l'état des connaissances

Dans le cadre du projet de l'état des connaissances, le MPO prévoit publier douze articles de synthèse portant chacun sur un aspect des effets environnementaux de l'aquaculture. Le présent volume contient les trois articles suivants : Effets environnementaux à distance de la pisciculture marine, Effets écosystémiques de l'élevage de bivalves marins et Utilisation de produits chimiques en pisciculture marine au Canada : étude des pratiques actuelles et effets possibles sur l'environnement.

Renseignements supplémentaires

Pour de plus amples renseignements sur un article, veuillez communiquer avec son auteur principal. Pour de plus amples renseignements sur le projet de l'état des connaissances, veuillez communiquer avec :

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FAR-FIELD ENVIRONMENTAL EFFECTS OF MARINE FINFISH AQUACULTURE

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EXECUTIVE SUMMARY

This review evaluates the existing knowledge and research needs required to determine the ability of coastal waters to support a sustainable marine finfish aquaculture industry. A central question is what methods, environmental observations and models exist, or are required, to determine the capacity of coastal areas to assimilate additional sources of dissolved and particulate matter released by cultured finfish.

Pillay (1992) provided an early review of major environmental effects of all types of aquaculture on a worldwide basis. Over the past decade, several international groups have considered various environmental issues surrounding the development of marine finfish aquaculture (Rosenthal 1988, 1994; GESAMP 1991, 1996; Buerkly 1993; Stewart et al. 1993; Ervik et al. 1994a, 1997; Stewart 1994, 2001; Rosenthal et al. 1995; Silvert and Hargrave 1995; Burd 1997; Goldberg and Triplett 1997; Milewski et al. 1997; Fernandes et al. 2000; Harvey 2000; Milewski 2000; EVS 2001; Holmer et al. 2001). Much of the information on environment-finfish aquaculture interactions in publications cited above is focused on measurable near-field changes in water and sediment variables sensitive to organic matter and nutrient additions.

Despite the difficulties of observing far-field effects, published literature shows that in some locations, measurable effects attributable to finfish aquaculture development have been observed at the ecosystem level. The impacts may be categorized into three types of broad-scale changes distant from farm sites: eutrophication, sedimentation and effects on the food web.

It is a common observation that the amount of suspended particulate matter increases in the immediate vicinity of finfish net-pens. When feed pellets are distributed by hand or automatic mechanical feeders, a fine dust may potentially be transported in the air or trapped in the water surface film and spread over a broad area. Unconsumed feed pellets and fish feces usually contribute to increased local concentrations of suspended and sedimented particulate matter. While much of the released material is assumed to settle rapidly at or near cage sites (Gowen et al. 1994; Silvert 1994e; Findlay et al. 1995; Findlay and Watling 1997), there is potential for horizontal transport and widespread dispersion, particularly in areas with high currents (Sutherland et al. 2001; Cromey et al. 2002). Holmer (1991) collected material, directly attributable to a finfish aquaculture source, at distances up to 1.2 km from a farm site in Danish coastal water. The extent to which resuspension and lateral transport increase sedimentation at locations remote from farm sites depends on both physical and sedimentological processes. Tidal flow,

residual circulation, patterns of turbulence, wind and wave energy, and flocculation (aggregation) will determine large-scale patterns of particle dispersion. The distances and locations of accumulation are highly site-specific and depend on bottom topography, currents, erosion and flocculation processes that affect the residence time of material both in the column (Sutherland et al. 2001) and on the bottom (Milligan and Loring 1997).

Specific compounds associated with organic matter, such as fatty acids, digestible proteins, sterols, elemental sulfur, pristane and stable carbon/nitrogen isotopes (Li-Xun et al. 1991; Johnsen et al. 1993; Findlay et al. 1995; McGhie et al. 2000) and trace elements such as zinc that might be used as tracers of fish feed pellets, have been measured in surface sediments to determine far-field dispersion patterns (Ye et al. 1991; McGhie et al. 2000; Sutherland et al. 2002; Yeats 2002). Alteration of bottom type to more fine-grained sediments through enhanced deposition of flocculated, fine-grained material may also account for the speculation that a population of lobsters was displaced from their historic spawning ground after a salmon farm was located at the site (Lawton and Robichaud 1991). However, an opposite effect of salmon farm operations causing aggregations of lobster may also occur. Salmon farm sites may be a refuge for lobsters from harvesting.

Eutrophication is the process of natural or anthropogenic enrichment of aquatic systems with inorganic nutrient elements (Jørgensen and Richardson 1996; Strain and Yeats 1999; Cloern 2001). Long-term eutrophication of coastal and estuarine waters results from the additions of both dissolved inorganic and organic nutrients and increased biological oxygen demand (BOD) from oxygen-consuming material from all sources (Rosenberg 1985; Costa-Pierce 1996; Johannessen and Dahl 1996; Cloern 2001). Dissolved inorganic nutrients released by finfish culture and regenerated from sediments enriched with sedimented organic matter under fish pens may stimulate phytoplankton production and increase oxygen demand. It is often difficult to accurately estimate the magnitude of additions of nutrients and organic matter from finfish aquaculture when many environmental factors and possible sources of addition occur (Einen et al. 1995; Strain et al. 1995). Models can help determine the relative amounts of organic loading from aquaculture from all natural sources (river discharge, tidal exchange, rainfall, phytoplankton and macroalgal production) and human inputs (Valiela et al. 1997). The degree of nutrient enrichment is influenced by the scale of aquaculture, local hydrographic characteristics, the magnitude of other sources relative to aquaculture and internal processes, such as uptake by phytoplankton, algae, internal recycling, resuspension of fine material, and uptake by biofouling communities that colonize net-pens.

The effects of eutrophication may extend into shallow water littoral and intertidal zones. Intertidal areas, subject to daily movements of water and sediment, are locations influenced by broad-scale processes affecting chemical fluxes of mass and dissolved material throughout an inlet system. Nutrient enrichment can stimulate the extensive development of macroalgal beds (Soulsby et al. 1982; Petrell et al. 1993; Campbell 2001), which have a large capacity for nutrient uptake (Chopin and Yarish 1999; Chopin *et al. 2000) and may affect benthic fauna through changes in the rates and nature of deposition of particulate organic matter (Bourget et al. 1994). However, few studies have unequivocally linked the establishment of aquaculture farm sites to environmental or ecological changes in intertidal areas.*

Eutrophication can alter the ratio between essential nutrients (carbon: nitrogen:phosphorus), as well as absolute concentrations by causing a shift in phytoplankton species assemblages. It has proven difficult to directly relate the occurrence of harmful algal blooms (HAB) to finfish farms. As with other types of plankton blooms, many environmental factors appear to control the formation of HABs. Water column mixing and stratification that maintain cells in the photic zone with an adequate nutrient supply are critical variables. In contrast to numerous studies of localized benthic effects of finfish aquaculture at farm sites, there have been very few observations of effects on plankton communities (Burd 1997). Reductions in zooplankton standing stock with oxygen depletion could allow standing stocks of phytoplankton to increase. With sufficient nutrient and light supplies, higher rates of primary production and increased sedimentation would result in even further oxygen depletion in deep water.

There is an extensive literature documenting changes in benthic infauna community structure associated with high levels of nutrient and organic matter additions (Burd 1997). Only fauna (e.g. nematodes and polychaetes) tolerant of low oxygen conditions and reduced sulfides are able to survive under conditions of high organic sedimentation (Hargrave et al. 1993, 1997; Duplisea and Hargrave 1996). The presence/absence of these 'indicator' species or faunal groups may show transitions from lower (background) levels of organic matter supply to high deposition rates caused by unconsumed feed pellets and fish feces in areas subject to low transport (Weston 1990; Pocklington et al. 1994; Burd 1997). Moderate increases in organic matter supply may stimulate macrofauna production and increase species diversity; however, with increasingly higher rates of organic input, diversity and biomass decrease.

Widespread changes in species community composition of benthic macrofauna distant from farm sites are more difficult to detect and have been less studied. Temporal and spatial scales of changes in benthic macrofauna species composition and biomass have been measured over the past decade in some areas as part of long-term monitoring programs near net-pens to determine if organic enrichment effects from aquaculture can be detected (Burd 1997; Brooks 2001). Most studies have shown that the local extent of altered benthic community structure and biomass is limited to less than 50 m. Water depth and current velocity are critical factors determining patterns of sedimentation around cage sites (Weston 1990; Pohle et al. 1994; Silvert 1994e; Henderson and Ross 1995; Burd 1997; Pohle and Frost 1997; Brooks 2001; Cromey et al. 2002), and therefore impacts of benthic fauna differ at different farm sites. In southwest New Brunswick, organic enrichment effects at newly established farm sites were localized to within 30 m of cages. After approximately five years, changes were measurable over greater (>200 m) distances. Macrofaunal community diversity was most reduced close to a farm site that had been in operation for 12 years, but significant declines in diversity also occurred throughout the inlet system. Benthic epifauna and infauna at two intertidal *sites at varying distances from aquaculture sites showed that the diversity of infauna was significantly higher away (>500 m) than near (<500 m) farm sites (Wong et al. 1999). Loss of diversity at distances less than 500 m may indicate that benthic infauna are more sensitive to organic matter additions than epifauna (Warwick 1986, 1987), possibly reflecting changes in sediment physical structure (grain size), oxygen supply and sulfide accumulation associated with increased organic matter supply.*

Another far-field effect of local sources of organic matter produced by finfish farm sites involves the use of chemotherapeutants. Antibiotics in medicated fish feed have the potential to induce drug resistance in natural microbial populations on an inlet-wide scale. Concentrations of a commonly used antibiotic, oxytetracycline (OTC), largely disappeared within a few weeks, but traces of the antibiotic were detectable for up to 18 months (Samuelsen et al. 1992). In Puget Sound, the highest numbers of bacteria (as colony-forming units) in sediments generally occurred at farm sites (Herwig et al. 1997), but the proportion of OTC resistant bacteria declined exponentially with increasing distance from a farm. Ervik et al. (1994b) also observed antibiotics in fish and wild mussels near a farm site after medicated food had been administered, and OTC resistance has been observed in bacteria cultured from sediments up to 100 m away from salmon farm sites in inlets in the Bay of Fundy where salmon farms are concentrated (Friars and Armstrong 2002).

GAPS IN KNOWLEDGE

1. *There is a need to determine sustainable levels of salmon production within coastal regions, inlets or embayments where marine finfish aquaculture is currently practiced in Canada.*

2. *Mass balance models of nutrient loading (inorganic and organic) from all sources (natural and anthropogenic) may be used to assess potential additions from finfish aquaculture. Budgets must take into account internal nutrient recycling, as well as external sources.*

3. *General circulation models can be developed and improved to resolve combined effects of tidal and wind-driven forcing and that reflect complex topography and intertidal drying zones.*

4. *New methods are required to quantify processes of resuspension that redistribute fine material produced locally by finfish aquaculture sites over large areas.*

5. *New methods are required to quantify processes, such as flocculation and aggregation, that affect dispersion of particulate matter from finfish farm sites.*

6. *Studies are required to determine if the frequency and location of HABs or plankton blooms are related to the expansion of finfish aquaculture.*

7. *New studies are required to determine changes in water column variables in areas of intensive finfish aquaculture. In comparison to benthic studies, there have been very few investigations of changes in planktonic communities around finfish aquaculture sites.*

8. *Further studies are required to document environmental or ecological changes in intertidal areas and to determine if these can be linked unequivocally to the establishment of aquaculture sites.*

9. *Mass balance and numerical models are required to link production and external loading with aerobic and anaerobic oxidation of organic matter (pelagic and benthic), sedimentation and sulfide accumulation in sediment.*

10. *Further studies are required to determine the extent of far-field effects on ecological and biological impacts of antibiotic resistance induced in microbial and other wild populations in areas of intensive finfish aquaculture.*

EFFETS ENVIRONNEMENTAUX À DISTANCE DE LA PISCICULTURE MARINE

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RÉSUMÉ

Ce document évalue les connaissances actuelles et les recherches nécessaires pour déterminer la capacité des eaux côtières à soutenir une industrie de pisciculture marine durable. Une question centrale est de savoir quels méthodes, observations environnementales et modèles existent ou sont requis pour déterminer la capacité de régions côtières à assimiler les apports supplémentaires de matières dissoutes ou particulaires provenant des piscicultures.

Pillay (1992) a passé en revue les principaux effets environnementaux de tous les types d'aquaculture à l'échelle mondiale. Depuis une décennie, plusieurs groupes internationaux ont abordé divers enjeux environnementaux liés au développement de la pisciculture marine (Rosenthal, 1988 et 1994; GESAMP, 1991 et 1996; Buerkly, 1993; Stewart et al., 1993; Ervik et al., 1994a et 1997; Stewart, 1994 et 2001; Rosenthal et al., 1995; Silvert et Hargrave, 1995; Burd, 1997; Goldberg et Triplett, 1997; Milewski et al., 1997; Fernandes et al., 2000; Harvey, 2000; Milewski, 2000; EVS, 2001; Holmer et al., 2001). Une bonne partie de l'information présentée dans ces publications sur les interactions entre la pisciculture et l'environnement porte sur les modifications mesurables à courte distance de variables, liées à l'eau et aux sédiments, sensibles à des apports supplémentaires de matière organique et d'éléments nutritifs.

Malgré la difficulté d'observer les effets à distance, la littérature sur le sujet indique que des effets écosystémiques du développement de la pisciculture marine ont été mesurés à certains endroits. Ces impacts à distance peuvent être rangés en trois catégories : eutrophisation, sédimentation et effets sur le réseau trophique.

On observe couramment une quantité accrue de particules en suspension à proximité immédiate des enclos à poissons. Lorsque des granules de nourriture sont distribués manuellement ou par des distributeurs automatiques, une fine poussière peut être transportée dans l'air ou piégée à la surface de l'eau et s'étendre sur une grande superficie. Les granules de nourriture non consommés et les excréments de poisson entraînent habituellement une hausse des concentrations locales de particules en suspension ou déposées au fond. Bien qu'on suppose qu'une bonne partie de cette matière se dépose rapidement près des enclos d'élevage (Gowen et al., 1994; Silvert, 1994e; Findlay et al., 1995; Findlay et Watling, 1997), elle peut être transportée horizontalement et se disperser considérablement, particulièrement là où il y a de forts courants (Sutherland et al., 2001; Cromey et al., 2002). Dans les eaux côtières du Danemark, Holmer (1991) a échantillonné de la matière provenant directement d'une

pisciculture jusqu'à 1,2 km de celle-ci. La mesure dans laquelle la remise en suspension et le transport latéral accroissent la sédimentation à des endroits éloignés des sites piscicoles dépend de processus physiques et sédimentologiques. Les courants de marée, la circulation résiduelle, les régimes de turbulence, l'énergie des vagues et du vent ainsi que la floculation (formation d'agrégats) déterminent la dispersion des particules à grande échelle. Les distances et les emplacements des accumulations sont propres à chaque site et dépendent de la topographie du fond, des courants et des processus d'érosion et de floculation qui influent sur le temps de séjour de la matière tant dans la colonne d'eau (Sutherland et al., 2001) qu'au fond (Milligan et Loring, 1997).

Des composés précis associés à la matière organique, comme des acides gras, des protéines digestibles, des stérols, le soufre élémentaire, le pristane, les isotopes stables du carbone et de l'azote (Li-Xun et al., 1991; Johnsen et al., 1993; Findlay et al., 1995; McGhie et al., 2000) et des éléments traces, comme le zinc, pouvant servir de traceurs des granules de nourriture à poisson ont été mesurés dans des sédiments de surface pour déterminer la dispersion à distance (Ye et al., 1991; McGhie et al., 2000; Sutherland et al., 2002; Yeats 2002). La modification du type de fond par le dépôt accru de matière fine floculée, donnant un sédiment plus fin, pourrait expliquer l'hypothèse voulant que l'établissement d'une salmoniculture ait entraîné l'abandon d'une frayère par une population de homards (Lawton et Robichaud 1991). Toutefois, les salmonicultures peuvent aussi avoir un effet inverse, soit d'occasionner des concentrations de homards. Les sites salmonicoles pourraient offrir aux homards un refuge contre leur exploitation.

L'eutrophisation est le processus d'enrichissement naturel ou anthropique d'écosystèmes aquatiques en éléments nutritifs inorganiques (Jørgensen et Richardson, 1996; Strain et Yeats, 1999; Cloern, 2001). L'eutrophisation à long terme de milieux estuariens ou côtiers découle de l'apport de substances nutritives dissoutes organiques ou inorganiques et d'une demande biologique en oxygène (DBO) accrue due à la décomposition de matière de toutes les sources (Rosenberg, 1985; Costa-Pierce, 1996; Johannessen et Dahl, 1996; Cloern, 2001). Des éléments nutritifs inorganiques dissous provenant de piscicultures ou régénérés à partir de sédiments enrichis en matière organique sédimentée sous les enclos à poisson peuvent stimuler la production phytoplanctonique et accroître la demande en oxygène. Il est souvent difficile d'estimer exactement l'apport d'éléments nutritifs et de matière organique provenant d'une pisciculture lorsque les sources possibles et les facteurs environnementaux sont nombreux (Einen et al., 1995; Strain et al., 1995). Des modèles peuvent aider à déterminer les quantités relatives d'apports organiques provenant de l'aquaculture et de sources naturelles (transport par les cours d'eau ou les marées, précipitations et production phytoplanctonique et macroalgale) ou anthropiques (Valiela et al., 1997). Le niveau d'enrichissement en éléments nutritifs dépend de l'ampleur de l'exploitation aquacole, des caractéristiques hydrographiques locales, de l'ampleur des autres sources par rapport à l'aquaculture et de processus internes, comme l'absorption par les algues phytoplanctoniques ou autres, le recyclage interne, la remise en suspension de matière fine et l'absorption par les communautés salissantes qui colonisent les enclos.

Les effets de l'eutrophisation peuvent s'étendre aux zones littorales ou intertidales peu profondes. Soumises à des mouvements quotidiens d'eau et de sédiment, les zones intertidales subissent l'influence de processus à grande échelle qui régissent les flux de matières particulaires et dissoutes dans un réseau d'échancrures de la côte. L'enrichissement en éléments nutritifs peut favoriser le développement de grands herbiers de macroalgues (Soulsby et al., 1982; Petrell et al., 1993; Campbell, 2001), qui ont une forte capacité d'absorption d'éléments nutritifs (Chopin et Yarish, 1999; Chopin et al., 2000) et peuvent influer sur la faune benthique en modifiant les taux de sédimentation et la nature de la matière organique particulaire déposée (Bourget et al., 1994). Toutefois, peu d'études ont établi un lien sans équivoque entre la création d'exploitations aquacoles et des changements environnementaux ou écologiques en zone intertidale.

L'eutrophisation peut changer le rapport entre les éléments nutritifs essentiels (carbone, azote et phosphore) ainsi que leurs concentrations absolues en modifiant les assemblages d'espèces de phytoplancton. Il est difficile de relier directement des cas de prolifération d'algues nuisibles à des piscicultures. Comme pour d'autres types de prolifération de phytoplancton, de nombreux facteurs environnementaux semblent déterminer les proliférations d'algues nuisibles. Le mélange et la stratification de la colonne d'eau qui maintiennent les cellules dans la zone photique avec suffisamment d'éléments nutritifs constituent des variables essentielles. Contrairement à de nombreuses études qui ont mis en évidence des effets de piscicultures sur les communautés benthiques à proximité, très peu d'effets ont été observés sur les communautés planctoniques (Burd, 1997). Des réductions de la biomasse du zooplancton attribuables à l'épuisement de l'oxygène pourraient permettre l'accroissement de la biomasse du phytoplancton. Avec suffisamment de lumière et d'éléments nutritifs, des taux accrus de production primaire et de sédimentation épuiseraient encore davantage l'oxygène en eau profonde.

Il existe une littérature volumineuse qui rapporte des modifications de la structure de la communauté endobenthique associées à des apports élevés d'éléments nutritifs et de matière organique (Burd, 1997). Seuls les animaux qui tolèrent de faibles concentrations d'oxygène et la présence de sulfures réduits (p. ex., nématodes et polychètes) peuvent survivre dans des conditions de forte sédimentation organique (Hargrave et al., 1993 et 1997; Duplisea et Hargrave, 1996). La présence ou l'absence de ces espèces ou groupes d'espèces « indicatrices » peut montrer la transition d'un niveau faible (naturel) d'apport de matière organique à un taux de sédimentation élevé causé par des granules de nourriture non consommés et des excréments de poisson dans des zones soumises à un faible transport (Weston, 1990; Pocklington et al., 1994; Burd, 1997). Des hausses modérées de l'apport de matière organique peuvent stimuler la production macrofaunique et accroître la diversité en espèces, mais, au-delà d'un certain seuil, l'augmentation de l'apport organique entraîne une baisse de la diversité et de la biomasse.

Les modifications à grande échelle de la composition spécifique de la communauté macrobenthique à distance des sites piscicoles sont plus difficiles à déceler et n'ont pas été beaucoup étudiées. Depuis une décennie, dans le cadre de programmes de *surveillance à long terme, on mesure dans certains secteurs les échelles temporelles et spatiales des modifications de la composition spécifique et de la biomasse de la communauté macrobenthique, afin de déterminer si l'on peut détecter des effets d'enrichissement en matière organique attribuables à l'aquaculture (Burd, 1997; Brooks, 2001). La plupart des études ont montré que ces modifications ne s'étendent pas au-delà de 50 m des sites piscicoles. Comme la profondeur de l'eau et la vitesse des courants sont des facteurs déterminants de la sédimentation près des enclos (Weston, 1990; Pohle et al., 1994; Silvert, 1994e; Henderson et Ross, 1995; Burd, 1997; Pohle et Frost, 1997; Brooks, 2001; Cromey et al., 2002), les impacts sur la faune benthique varient selon les sites piscicoles. Dans le sud-ouest du Nouveau-Brunswick, les effets d'enrichissement en matière organique attribuables à des piscicultures nouvellement établies ne dépassaient pas une distance de 30 m des enclos. Après environ cinq ans, des modifications pouvaient être mesurées à de plus grandes distances (> 200 m). La plus grande réduction de la diversité macrobenthique a été observée près d'une pisciculture exploitée depuis douze ans, mais des baisses significatives de la diversité se sont également produites dans l'ensemble du réseau d'échancrures de la côte. Dans une étude de l'épifaune et de l'endofaune benthiques à deux endroits situés à différentes distances d'exploitations aquacoles, Wong et al. (1999) ont montré que la diversité de l'endofaune était significativement plus élevée à distance (> 500 m) qu'à proximité (< 500 m) de pisciculture. La diversité réduite à des distances inférieures à 500 m pourrait indiquer que la faune endobenthique est plus sensible que la faune épibenthique aux apports de matière organique (Warwick, 1986 et 1987), peut-être en réaction aux modifications de la structure physique des sédiments (granulométrie), de la concentration d'oxygène et de l'accumulation de sulfure associées à des apports accrus de matière organique.*

Un autre effet à distance des piscicultures découle de l'utilisation d'agents chimiothérapeutiques. Les antibiotiques ajoutés à la nourriture du poisson peuvent entraîner une pharmacorésistance chez les populations microbiennes naturelles à l'échelle d'une échancrure de la côte. Après quelques semaines, les concentrations d'oxytétracycline (OTC), un antibiotique d'utilisation courante, ont considérablement diminué, mais des traces de l'antibiotique étaient détectables pour une période allant jusqu'à 18 mois (Samuelsen et al., 1992). Dans Puget Sound, on trouve généralement les sédiments contenant le plus de bactéries (unités formant colonies) aux sites piscicoles (Herwig et al., 1997), mais la proportion de bactéries résistantes à l'OTC diminue exponentiellement à mesure que l'on s'éloigne d'une pisciculture. Ervik et al. (1994b) ont aussi observé la présence d'antibiotiques chez des poissons et des moules sauvages à proximité d'une pisciculture où l'on avait ajouté des antibiotiques à la nourriture du poisson. Friars et Armstrong (2002) ont mis en évidence une résistance à l'OTC chez des bactéries cultivées à partir de sédiments situés jusqu'à 100 m de salmonicultures dans des échancrures de la baie de Fundy.

LACUNES DANS LES CONNAISSANCES

1. *Il faut déterminer les niveaux durables de production salmonicole dans les régions côtières où l'on pratique actuellement la pisciculture marine au Canada.*

2. *Des modèles de bilan de masse des apports de substances nutritives (inorganiques et organiques) provenant de toutes les sources (naturelles et anthropiques) peuvent servir à évaluer les apports éventuels de la pisciculture. Ces bilans doivent prendre en compte le recyclage interne des éléments nutritifs en plus des sources externes.*

3. *On peut élaborer ou améliorer des modèles de circulation générale de l'eau pour déterminer les effets combinés des forçages par les marées et le vent en tenant compte de la complexité de la topographie et des zones intertidales.*

4. *Il faut de nouvelles méthodes pour quantifier les processus de remise en suspension qui redistribuent sur de grandes superficies la matière fine provenant des piscicultures.*

5. *Il faut de nouvelles méthodes pour quantifier les processus, comme la floculation et l'agrégation, qui influent sur la dispersion des particules provenant des piscicultures.*

6. *Il faut réaliser des études pour déterminer si la fréquence et les emplacements de proliférations d'algues nuisibles ou non sont liés à l'expansion de la pisciculture.*

7. *Il faut effectuer de nouvelles études pour détecter les modifications touchant les variables liées à la colonne d'eau dans les régions de pisciculture intense. Les modifications des communautés planctoniques près des piscicultures ont fait l'objet de beaucoup moins d'études que celles touchant les communautés benthiques.*

8. *Il faut réaliser des études approfondies pour mettre en évidence des changements environnementaux ou écologiques dans les zones intertidales et relier catégoriquement ces modifications à la création d'exploitations aquacoles.*

9. *Il faut des modèles numériques, notamment de bilan de masse, pour relier la production et les apports externes avec l'oxydation aérobie et anaérobie de la matière organique (pélagique et benthique), la sédimentation et l'accumulation de sulfure dans les sédiments.*

10. *Il faut réaliser des études approfondies pour déterminer l'ampleur des effets biologiques et écologiques à distance de la résistance aux antibiotiques des populations microbiennes sauvages dans les régions de pisciculture intensive.*

INTRODUCTION

This paper on far-field environmental effects of marine finfish aquaculture reviews existing knowledge and research needs required to determine the ability of coastal waters to support a sustainable marine finfish aquaculture industry. A central question is what methods, observations and models exist, or are required, to determine the capacity of coastal areas to assimilate additional sources of dissolved and particulate matter released by cultured finfish. Related topics of diseases in cultured fish and transmission to other species, genetic interactions, displacement of wild salmon stocks through escapement, and impacts of chemicals and therapeutants used in the industry will be dealt with in other papers in this series.

EXISTING KNOWLEDGE – MAIN ISSUES

Pillay (1992) provided an early review of major environmental impacts of all types of aquaculture on a worldwide basis. More recently, effects of fish cage culture have been reviewed for various marine species and environments (Black 2001). Over the past decade, several international groups have considered various environmental issues surrounding the development of marine finfish aquaculture, which involve many of the potential far-field environmental and ecological effects discussed below.

The Environmental Interactions of Mariculture Working Group was established under the ICES Mariculture Committee in 1988, and several review articles have been produced (Rosenthal 1988, 1994; Rosenthal et al. 1995). An integrated approach to considering finfish aquaculture within a broad coastal zone management system has been developed in Norway (MOM and LENKA Projects) (Stewart et al. 1993; Ervik et al. 1997; Stewart 2001). A joint Canada/Norway workshop on environmental impacts of aquaculture was held in Bergen, Norway (February 1993) (Ervik et al. 1994a; Stewart 1994). The ICES Mariculture Working Group sponsored a modelling workshop on the same topic (Silvert and Hargrave 1995), and papers presented at an ICES international symposium in 1999 were published in the ICES Journal of Marine Science (Holmer et al. 2001).

The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) produced three reports (GESAMP 1991, 1996, 2001) on monitoring effects and environmental impacts of finfish aquaculture within the context of a management framework for coastal development. Burd (1997) reviewed published literature on environmental interactions of salmon aquaculture for the British Columbia (BC) Environmental Assessment Office. Similar reviews have been prepared for the Conservation Council of New Brunswick (Buerkly 1993; Milewski et al. 1997; Harvey 2000) and the Environmental Defense Fund (Goldberg and Triplett 1997) and presented recently at a SeaWeb conference on Marine Aquaculture and the Environment (Milewski 2000; Tlusty et al. 2000a). A review of selected literature on impacts of freshwater and marine aquaculture on the environment was completed in 2000 for Fisheries and Oceans Canada (DFO) (EVS 2001). Environmental monitoring programs and regulations for marine aquaculture in various countries have been reviewed in a special issue of Journal of Applied Ichthyology (2000: Volume 16). Fernandes et al. (2000) and other contributors described progress towards establishment of scientific guidelines for Best Environmental Practice in individual countries, such as 'MARAQUA' Concerted Action - an attempt to achieve standards for aquaculture development and environmental protection among European countries. Finally, a review focused on eutrophication effects of salmon aquaculture in Scottish marine waters was compiled by Black (2003).

Much of the information on environment-finfish aquaculture interactions in publications cited above is focused on measurable near-field changes in water and sediment variables sensitive to organic matter and nutrient additions. Factors such as decreased dissolved oxygen, increased dissolved nutrients and suspended particulate organic matter, increased organic sedimentation, and associated changes in benthic habitat and communities are the most commonly observed environmental changes (Burd 1997). Increases in dissolved nutrients and organic matter loading are directly related to the magnitude of fish biomass held in pens and may be modelled based on knowledge of a few variables (number and size of fish, water depth, temperature and feeding rate) (Silvert 1992, 1994a; Silvert and Sowles 1996; Silvert and Cromey 2001). In areas where currents are low (e.g. $<5\ cm \cdot s^{-1}$), direct effects of increased particulate matter sedimentation have been measured under and close to (<50 m) cage sites, when compared to sedimentation rates measured some distance (>500 m) away (Sutherland et al. 2001). Reductions in benthic faunal diversity, increased community metabolism and creation of anoxic conditions have been quantitatively related to organic matter supply (Kupka-Hansen et al. 1990; Weston 1990; Hargrave et al. 1993, 1997; Findlay et al. 1995; Findlay and Watling 1997). All of these effects are diminished or are not observed in areas of high currents.

Currents (direction, velocity and variability), depth and substrate type are three major factors determining both local and far-field effects of finfish aquaculture in coastal areas (Findlay et al. 1995; Burd 1997). These variables determine the spatial and temporal heterogeneity of coastal marine environments that makes prediction and observation of far-field effects potentially caused by aquaculture development difficult to observe and quantify. Near-shore marine environments are among the most physically, chemically and biologically diverse marine habitats in the world (Valiela 1991). For example, bathymetric and hydrographic variables, including the input of tidal and wind energy, determine patterns of water circulation and the nature of bottom substrates in all coastal areas. Thus, changes in variables become more difficult to measure with increasing distance from farm sites. The horizontal extent of broad-scale ecosystem effects that might be expected to occur due to finfish aquaculture will be highly site-specific and dependent on the size, spacing and stocking density of net-pens. The ability to observe these effects is also confounded by other inputs of nutrient and particulate matter from urban and industrial sources.

Despite the difficulties of observing far-field effects, published literature shows that in some locations, measurable effects attributable to finfish aquaculture development have been observed at the ecosystem level (Pearson and Black 2001). The impacts may be categorized into three types of broad-scale changes distant from farm sites:

1) **eutrophication** (nutrient and organic matter enrichment leading to increased dissolved inorganic nutrient concentrations and decreased dissolved oxygen);
2) **sedimentation** (direct settling of feed pellets, feces, increased organic matter flocculation leading to higher deposition rates of small particles, changes in turbidity);
3) **food web structure and function** (changes in planktonic, fish and benthic community composition, enhanced/decreased productivity, behavioral avoidance, stimulation of harmful algal blooms, intertidal/macroalgal community effects).

EUTROPHICATION

MODELLING ADDITIONS OF NUTRIENTS AND ORGANIC MATTER

Eutrophication is the process of natural or anthropogenic enrichment of aquatic systems with inorganic nutrient elements (Jørgensen and Richardson 1996; Strain and Yeats 1999; Cloern 2001). Nixon (1995) proposed the terms *oligotrophic, mesotrophic, eutrophic* and *hypertrophic* to describe the progression from low to high nutrient status. Long-term eutrophication of coastal and estuarine waters is due to additions of both dissolved inorganic and organic nutrients and increased biological oxygen demand (BOD) from oxygen-consuming material from all sources (Rosenberg 1985; Costa-Pierce 1996; Johannessen and Dahl 1996; Cloern 2001). Trophic status may also be measurable in sediments using biochemical variables related to the quantity and composition of organic matter (Dell'Anno et al. 2002). However, it is often difficult to accurately estimate anthropogenic impacts due to nutrient and organic matter additions from only one source, for example finfish aquaculture, when many possible sources exist (Einen et al. 1995; Strain et al. 1995). Few models take into account internal (recycling), as well as all external sources of nutrient loading in coastal locations (Burd 1997; Strain and Yeats 1999). Resuspension of fine material during storms may bring recently settled material and nutrient-rich sediment pore water into suspension. Phytoplankton production may be stimulated by regenerated nutrients to levels in excess of what would be supported by nutrient concentrations in the water column.

Eutrophic effects of nutrients released from fish farms have been documented and modelled in many countries for coastal waters with restricted exchange to offshore areas: Scotland (Gowen et al. 1988; Pridmore and Rutherford 1992; Ross et al. 1993, 1994; Gowen 1994; Berry 1996; Gillibrand and Turrell 1997; Black 2003), Sweden (Håkanson et al. 1988; Ackefors and Enell 1990; Mäkinen 1991; Enell and Ackefors 1992) and Norway (Ervik et al. 1985). Knowledge that the spread of disease between farms can occur over distances of several kilometers has also been incorporated into the development of an environmental monitoring program in Norway (MOM) (Stewart 1991, 1998; Ervik et al. 1997; Fisheries Research Services 2000; Hansen et al. 2001). The integrated management approach recognizes the need to regulate local environmental conditions at farm sites to ensure protection of a larger area (Ervik et al. 1997). Geographical Information Systems (GIS) are increasingly being proposed as a useful tool for siting decisions (Nath et al. 2000).

Silvert and Cromey (2001) reviewed various approaches to modelling broad-scale environmental effects of finfish aquaculture. Effects of nutrient loading from fish cages on coastal inlet systems have been modelled using ecometric analysis (Håkanson and Wallin 1990; Wallin and Håkanson 1991; Nordvarg and Håkanson 2002). The analysis produces load diagrams or empirical relationships between dose (e.g. nutrient input) and some measurable environmental effect (e.g. changes in water transparency related to increasing suspended matter concentrations or dissolved nutrient concentrations). Expert (decision-support) (DSS) computer-based systems have been developed to incorporate physical and biological variables associated with finfish farm operations for prediction of benthic organic matter loading rates and evaluation of environmental criteria for optimal siting (Silvert 1994b,c,d; Hargrave et al. 1996; Silvert and Sowles 1996; Doucette and Hargrave 2002; Hargrave 2002). Trophodynamic models have also been structured into a DSS format to use water, sediment and higher trophic level variables to develop an index of estuarine environmental quality (Ferreira 2000). Models and empirical relationships in most DSS applications are based on functional relations that are site-specific and to be valid must be used within the same range of empirical values from which regressions used for calculations were derived.

Other modelling studies have used simple box models to assess the relative potential for environmental effects of new and existing fish farm sites (Strain et al. 1995; Gillibrand and Turrell 1997; Black 2003). The models for various Scottish sea lochs, based solely on surface water flushing driven by tidal exchange, without consideration of effects of complex topography or exchange of surface and deep water, showed the potential for enhanced nutrient levels due to farm activity (termed the equilibrium concentration enhancement). The upper 5-m surface layer in several of 63 sea lochs modelled showed enhancement ratios for potential enrichment by dissolved ammonia of 2 to 13 times.

In contrast, Burd (1997) concluded that total dissolved nitrogen produced by the salmon aquaculture industry in BC was <0.3% of the nitrogen input to the Strait of Georgia and hence would be unlikely to significantly alter natural concentrations. Tidal transport and mixing in the Strait of Georgia may dilute local contributions from a point source such as a single finfish farm site. A similar situation may exist within the Broughton Archipelago region of south-central BC, where large-scale water mass characteristics are determined by exchanges between inshore and offshore waters (Stucchi et al. 2002). More limited exchange occurs between inshore and offshore areas within the Southwestern Isles Region of the Bay of Fundy, New Brunswick (SWNB) (Greenberg et al. 1997), where spatial variations in phytoplankton populations appear to reflect estuarine and offshore source areas (Page et al. 2002).

The importance of local hydrographic conditions in determining nutrient enrichment effects was demonstrated in a small coastal embayment in New Zealand with restricted water exchange. Salmon aquaculture was estimated to have increased nitrogen concentrations by 30% with a corresponding increase in phytoplankton primary production (Pridmore and Rutherford 1992). The degree of nutrient enrichment due to releases from aquaculture sites will be different in different areas. This will be determined in part by the scale of aquaculture relative to local hydrographic characteristics (water residence time and inshore-offshore exchanges), the magnitude of other sources relative to aquaculture, and internal processes, such as uptake by phytoplankton and attached macro- and microalgae that remove nutrients.

Dissolved inorganic nutrient concentrations in macrotidal areas of the Bay of Fundy like Letang Inlet (SWNB) may be naturally elevated due to riverine input, sediment resuspension and recycling. When combined with inputs from industrial sources (such as fish processing plants and sewage discharge), effects of nutrients released by numerous fish farms may only be observed in regions where farm sites are highly concentrated (Strain et al. 1995). Nutrient enrichment may be less important in deep water coastal areas of Newfoundland-Labrador (NL) and BC, where farms tend to be more widely dispersed.

Dissolved ammonium concentrations in Letang Inlet (SWNB) during the summer are currently (1999/2000) about twice the levels observed in other Atlantic coastal inlets where salmon aquaculture is not carried out (Bugden et al. 2001). This observation in itself indicates that supply of this essential nutrient for phytoplankton production exceeds removal rates. In most temperate coastal areas that do not receive urban or industrial effluents, essentially all dissolved inorganic nitrogen is removed from the photic zone by phytoplankton during the summer growth period. However, it is possible that light, rather than nutrient supply, limits primary production in the SWNB area and elsewhere in the relatively turbid waters of the Bay of Fundy (Harrison and Perry 2001). Increasing dissolved nutrients will not stimulate higher levels of primary production if light-limiting conditions occur (Cloern 2001). It must also be emphasized that while nitrogen is often considered essential for phytoplankton production, other nutrients (e.g. phosphorus, silicate) may at times be limiting. The ratio between essential nutrients (carbon: nitrogen:phosphorus), as well as absolute concentrations, may be altered through the process of eutrophication causing a shift in phytoplankton species assemblages (Black 2003).

MODELLING DISTRIBUTIONS OF DISSOLVED OXYGEN AND BIOLOGICAL OXYGEN DEMAND

Models can be used to compare the relative amounts of dissolved nutrient and organic matter loading from aquaculture with natural sources (river discharge, tidal exchange, rainfall, phytoplankton and macroalgal production) and human inputs (Aure and Stigebrandt 1990; Valiela et al. 1997). Strain et al. (1995) compared estimated annual oxygen demand from salmon aquaculture with other natural and industrial sources of BOD in Letang Inlet to show that inputs of carbon, nitrogen, phosphorus and BOD from finfish aquaculture were of a similar magnitude or exceeded estimated phytoplankton production. It was formerly believed that coastal and estuarine areas of a macrotidal environment like the Bay of Fundy would experience a magnitude of water exchange sufficient to dilute and disperse local sources of organic matter arising from finfish farm sites. However, modelling of M_2 tidal circulation in SWNB has shown that water masses in some parts of the Letang Inlet system have complex and variable exchange with offshore areas (Greenberg et al. 1997, 2001; Dowd et al. 2001).

One indication that natural limits of organic matter supply have been exceeded by additional inputs is the formation of oxygen deficient water (GESAMP 2001). Oxygen is supplied to surface waters through advection of oxygenated offshore water, wind mixing and phytoplankton production during the day. However, these processes of supply are balanced by phytoplankton respiration, oxygen consumption by other planktonic organisms, and uptake by subtidal and intertidal sediments. Intensive finfish aquaculture also adds dissolved and particulate organic matter that is decomposed by bacteria to contribute additional oxygen demand. Dissolved inorganic nutrients released by finfish and regenerated from sediments enriched with sedimented organic matter under fish pens (discussed below) may stimulate phytoplankton production and hence indirectly increase oxygen demand (Aure and Stigebrandt 1990).

Observations of dissolved oxygen and nutrients over tidal cycles in the Letang area, SWNB near farm and reference locations showed that, at some times during summer months, concentrations were reduced in the immediate vicinity of net-pens (Wildish et al. 1993). Dissolved oxygen was close to 100% saturation during spring and early summer months as expected when microalgal production would add oxygen to the water during the day. However, an earlier study in Letang Harbour (Wildish et al. 1986a) showed that concentrations as low as 60% of saturation values occurred at some stages of a tidal cycle at locations >500 m from aquaculture sites. This is below a level (~80% saturation; approximately 7 $mg{\cdot}L^{-1}$) that would be stressful to fish (Davis 1975). Seasonal data for dissolved oxygen in Lime Kiln Bay, SWNB in 1991 and 1992 also showed that oxygen concentrations decreased from high values in the spring (>100% saturation) to minima in late summer and fall (80-90% saturation) (Martin et al. 1995; Page and Martin 2001). If stocking densities of cultured fish are high and located in areas where inlet-wide seasonal oxygen depletion occurs naturally, local oxygen levels within net-pens and near farm sites may be reduced below critical levels for sustained finfish aquaculture. A tidal circulation model developed for the Quoddy region (Greenberg et al. 2001) has been used to predict locations where critical combinations of farm location, current speed and duration could lead to oxygen deficiency due to respiration of cultured salmon (Page et al. 2002).

As the cited literature illustrates, most previous studies of effects of dissolved components released by finfish aquaculture have focused on dissolved inorganic nutrients and oxygen. Dissolved organic matter in seawater from diverse sources (released by phytoplankton and macrophytes, river input, runoff, sewage and aquaculture) is present in concentrations up to ten times greater than is present in particulate form (Kepkay and Bugden 2001). Seasonal changes, with order of magnitude increased concentrations during summer months, were correlated with changes in suspended chlorophyll in SWNB, indicating large seasonal variations in carbon loading (Kepkay et al. 2002). Inputs from aquaculture may be difficult to detect in the presence of such large natural variations. However, dissolved organic material is released in excretory products by cultured finfish, and these compounds have the ability to alter physiological responses in phytoplankton and other species. Arzul et al. (2001) examined growth of different phytoplankton species under experimental conditions in the presence of excretion products from various fish and shellfish. Growth inhibition with exposure to finfish excretory products was thought to be due to dissolved organic components or altered nitrogen:phosphorus ratios. The applicability of these results to natural waters, where excretory products produced by intensive finfish culture can be rapidly diluted, has not been determined.

ALGAL BLOOMS AND MACROPHYTE PRODUCTION

Previous research on harmful algal blooms (HAB) has shown that water column stratification and nutrient availability are critical variables for bloom formation. However, it has proven difficult to directly relate the occurrence of HABs to the establishment of finfish aquaculture farms (Page et al. 2001, 2002; Black 2003). In many areas, blooms of planktonic organisms that cause mortality to salmon have a historic occurrence unrelated to the development of finfish aquaculture (Stewart and Subba Rao 1996; Martin et al. 1999). Even small numbers of specific cell types (e.g. 5 cells$\cdot$mL^{-1} of *Chaetoceros sp.*) can cause gill damage in salmon and create portals for entry of infectious bacteria (Albright et al. 1993).

A long-term (10-yr) plankton monitoring program has been carried out in SWNB comparing sites near and distant from aquaculture operations (Martin and LeGresley 2001; Page et al. 2001). The observations show a trend to increased abundance of a diatom species (*Thalassiosira nordenskioeldii*) at sites near fish farms that might be correlated with aquaculture activity, but other environmental factors such as salinity also vary (Smith et al. 2001). As with other types of plankton blooms, many environmental factors appear to control the formation of HABs. While it is difficult to generalize, water column mixing and stratification that maintain cells in the photic zone with an adequate nutrient supply are critical variables. Eutrophic conditions of warmer water, increased nutrient levels and water column stability that favor the formation of all algal blooms generally contribute to poor culture conditions for fish such as Atlantic salmon (Stewart 1997).

Eutrophication effects may also extend into the shallow water littoral and intertidal zones where nutrient enrichment can stimulate the extensive development of macroalgal beds (Soulsby et al. 1982; Petrell et al. 1993; Campbell 2001). Macroalgae have a large capacity for nutrient uptake (Chapman and Craigie 1977; Subandar et al. 1993; Okhyun et al. 1998; Chopin and Yarish 1999; Chopin et al. 2000), and some species have alternative strategies for nutrient utilization (Anderson et al. 1981). Growth, biomass and chlorophyll pigment content of the green algae *Cladophora glomerata* was observed to have increased at sites close to fish farms in shallow (0.2-0.5 m) coastal waters of the northwest Baltic (Ruokolahti 1988). Increases were thought to reflect eutrophication effects due to increased dissolved phosphorus and nitrogen released from aquaculture sites, which reached maximum levels in August. Biomass of *Cladophora* throughout the northern Baltic also increased in the 1970s and 1980s, in response to general nutrient enrichment of northern regions of the Baltic Sea.

Biofouling communities that colonize nets or pens are also sites for extensive development of attached macroalgae. The biomass in these communities can be substantial (>1 kg·m^{-2}) (B. Hargrave, unpublished data from Bliss Harbour, SWNB), and nutrients released from fish and feces held within net-pens will be adsorbed by the biomass attached to the net surfaces. In systems like Bay d'Espoir, NL, the biofouling community is dominated by blue mussels that at times may constitute a significant biomass (M.R. Anderson, Northwest Atlantic Fisheries Centre, St. John's, NL, personal communication). Utilization of particulate wastes from cages by filter-feeding organisms such as mussels attached to net-pens has been examined (Stirling and Okumus 1995). Also, therapeutants such as antibiotics added to fish feed pellets were observed in nearby wild mussel populations (Ervik et al. 1994b), indicating transport and utilization of particulate matter released from farms where medicated feed had been used.

SEDIMENTATION

TRACERS OF DISPERSION

Specific compounds associated with fish feed pellets have been used as tracers of dispersion in surface sediments to determine far-field distribution patterns (Ye et al. 1991; McGhie et al. 2000; Brooks 2001; Sutherland et al. 2002; Yeats 2002). Variables such as fatty acids, elemental sulfur and pristane, indicative of enrichment by farm wastes, were measured from 100 to 150 m away from salmon pens at a site on the west coast of Norway (Johnsen et al. 1993). Similarly, increased concentrations in copper and zinc above natural background levels were observed in surface sediments away from farm sites in inlets in BC and SWNB where salmon aquaculture is most concentrated (Brooks 2001; Yeats 2002). Elevated zinc concentrations in under-cage sediments provided evidence that farms were the source of zinc away from farm sites. On the other hand, concentrations of fatty acids, sterols and digestible proteins in sediments near Swan's Island, Maine showed spatially limited effects of dispersion of farm-derived waste products (Findlay et al. 1995). Stable carbon isotopes were used to estimate that ~40% of sediment carbon up to 150 m from cages at a farm site in Tasmania was derived from organic wastes from fish pens (Li-Xun et al. 1991). Currents were relatively weak (2-5 cm·s^{-1}) in all locations where these studies were located; hence dispersion of settled waste may have been restricted. A study showing that fatty acids, sterols and stable carbon/nitrogen isotopes in sediments at a former net-pen site changed during fallowing means that these compounds may only be useful as tracers over short time scales (McGhie et al. 2000). They may have value to track near-field dispersion but be of limited use as far-field tracers.

Stable carbon isotopes were measured in suspended and sediment trap material in areas of the Broughton Archipelago, BC to determine transport of cage-derived particulate matter away from a farm site (Sutherland et al. 2001). The stable carbon (C) isotopic signature in feed pellets was detected in suspended matter within the net-pen system and in the near-field water column. However, although ^{13}C:^{12}C ratios in sediment trap material collected near the farm site consisted largely of fish fecal waste, they were not characteristic of feed. Metabolic conversion of feed consumed by fish and mixed sources of settled material (e.g. fish fecal waste mixed with natural marine seston and terrigenous organic debris) could have diluted the isotopic composition. Further studies such as this are warranted using naturally occurring isotopes to differentiate source material. For example, stable nitrogen isotopes have been used to show a progressive change in sources of organic matter supply to coastal and estuarine areas (McClelland and Valiela 1997). These methods may be used to detect distances and directions of particle transport produced through fish farming activity if there is a large difference in stable isotopic signatures in pen-derived and naturally occurring material (Li-Xun et al. 1991).

One problem in using tracers of net-pen generated material is that different chemical markers may show different patterns of dispersion. For trace metals such as copper and zinc it is necessary to normalize for background concentrations before enrichment can be quantified (Yeats 2002). Variability in dispersion patterns may also be due to the choice of tracer. Sutherland et al. (2002) observed variability and asymmetrical footprints for organic carbon, nitrogen and zinc in sediments under and away from net-pens in the Broughton Archipelago. Highest values were not always directly under pens, and the extent and shape of increased concentrations above background levels was different for each variable. Interactions between bathymetry, hydrography and sediment physical-chemical properties will determine the ultimate fate of material accumulated in bottom deposits both at and distant from farms. Site-specific studies are required to ground-truth patterns of dispersion for specific tracers.

PROCESSES CONTROLLING PARTICLE DISPERSION

The amount of suspended matter in surface water in the immediate vicinity of finfish net-pens is often increased over that measured some distance away (B. Hargrave, unpublished data from SWNB; Sutherland et al. 2001). Under calm conditions, there is also often an oil-rich, organic film surface visible at distances >1 km from farm sites. Particulate matter is easily dislodged from biofouling communities that colonize nets. Until the late 1990s, it was common practice in SWNB to remove biofouling communities from net-pens using high-pressure water jets on nets stretched over a large drum on a cleaning barge. Dislodged macrophyte and animal debris forming turbidity plumes carried by tidal currents have been observed at distances >1 km from a cleaning barge in SWNB (B. Hargrave, unpublished data).

When feed pellets are distributed by hand or automatic mechanical feeders, a fine dust may potentially be transported in the air or trapped in the water surface film and spread over a broad area. Unconsumed feed pellets and fish feces usually contribute to increased local concentrations of suspended and sedimented particulate matter. While much of the released material is assumed to settle rapidly (Gowen et al. 1994; Silvert 1994e), there is potential for horizontal transport and widespread dispersion, particularly in areas with high currents (Sutherland et al. 2001). In near-shore areas typical for finfish farm sites, water depth may be shallow (<30 m), and resuspended particles may cycle several times between the water column and sediment surface before reaching an area of low bottom shear stress and turbulence where permanent sedimentation occurs (Kranck et al. 1996a, b; Milligan and Loring 1997).

The extent to which resuspension and lateral transport increase sedimentation at locations remote from farm sites depends on both physical and sedimentological processes (Sutherland et al. 2001). Tidal flow, residual circulation, patterns of turbulence, and wind and wave energy will determine large-scale patterns of particle dispersion. Flocculation (aggregation) plays a major role by controlling the settling velocity of fine particles (Milligan and Loring 1997; Milligan et al. 2001). Material in suspension exists as either single grains or as part of flocs. Flocculation rate is primarily dependent on concentration and adhesion efficiency (the probability of two particles sticking together on contact). While turbulence initially favors aggregation by increasing the encounter rate of particles, above some level it causes floc break-up. Changes in the balance between these two modes of deposition, for example as a result of increased particle load, will directly affect settling velocity and hence particle dispersion.

Increased flocculation and sedimentation of particles that would normally remain in suspension have been observed in SWNB (Milligan and Loring 1997) and in estuaries in the United Kingdom (Ye et al. 1991). Flocculation and deposition of fine-grained sediments may be enhanced in coastal areas where dissolved organic matter (DOM) is added by natural sources (e.g. rivers containing humic substances), as well as by inputs from industrial and urban sources such as fish processing plants and sewage discharge. Colloidal material, the largest reservoir of organic matter in the ocean, creates aggregates of polysaccharide polymers that may enhance sedimentation and transfer of particles from the water column to the benthos (Kepkay 1994; Kepkay and Niven 1996). The development of finfish aquaculture in such areas creates an additional source of DOM that may then increase aggregate formation and deposition of material.

DIRECT OBSERVATIONS OF SEDIMENTATION

Most previous studies of increased sedimentation associated with marine finfish farm sites have focused on the presence of solid feed pellets and fish fecal waste in settled material at or near cage sites (Hargrave 1994; Findlay et al. 1995; Findlay and Watling 1997). Holmer (1991) collected material of this nature, directly attributable to a finfish aquaculture source, at distances up to 1.2 km from a farm site in Danish coastal waters. The distances and locations of accumulation are highly site-specific and depend on bottom topography, currents, erosion and flocculation processes that affect the residence time of material both in the column (Sutherland et al. 2001) and on the bottom (Milligan and Loring 1997). Two-dimensional tidally driven flow models coupled with particle tracking models have been used to predict changes in sedimentation and burial rates at variable distances around finfish farm sites in coastal areas of Maine (Dudley et al. 2000) and Scotland (Cromey et al. 2000, 2002). Results from applications of these models show that horizontal dispersion is sensitive to water depth, particle settling velocity, current velocity over depth and bottom shear stress at which settled pen-derived material may be resuspended. Improved understanding of processes involved with resuspension and erosion in both models allowed a range of sedimentation rates ($<$1-15 g $C \cdot m^{-2} \cdot day^{-1}$) around pens (up to 100 m) to be predicted that were consistent with measured values.

Burd (1997) and Brooks (2001) summarized extensive studies with short-term sediment traps deployed at distances up to 1 km from farm sites in coastal areas of BC and elsewhere. Interpretation of data from different studies is complicated by the fact that in some cases traps were placed directly on the seabed where resuspended sediment could increase apparent sedimentation rates. In an earlier comparison of similar data from different northern temperate latitude aquaculture sites, Hargrave (1994) concluded that a threshold (~1 g $C \cdot m^{-2} \cdot day^{-1}$) existed for sedimentation, such that at higher rates anoxic conditions were created in surface sediments. Oxygen supply was sufficient to maintain aerobic conditions at lower sedimentation rates in most locations. An earlier study under blue-mussel culture lines showed that similar sedimentation rates (above 1.7 g $C \cdot m^{-2} \cdot day^{-1}$) led to increased microbial sulfate reduction and sulfide accumulation (Dahlbäck and Gunnarsson 1981). The enhancement of anaerobic metabolism and formation of anoxic sediments with increased organic loading was similar to a relationship described by Sampou and Oviatt (1991) for a simulated eutrophication gradient with nutrients added to experimental mesocosms.

Findlay et al. (1995) and Findlay and Watling (1997) quantified relationships between carbon sedimentation and benthic response at finfish aquaculture sites in coastal Maine. They derived regression models linking benthic respiration (O_2 and CO_2 sediment-water exchange), organic matter sedimentation and oxygen supply (calculated from current velocity). Omori et al. (1994) also described a numerical model that linked aerobic and anaerobic oxidation of organic matter in sediments to sedimentation and sulfide accumulation. If sedimentation rates are known, these empirical relationships can be used to predict the degree of change expected in benthic community structure and metabolism with increasing rates of organic loading. Such relationships are included in the DEPOMOD model that predicts spatial patterns of deposition around farm sites (Cromey et al. 2000, 2002).

Rates of sedimentation at non-aquaculture sites are usually below the threshold (<1 g $C \cdot m^{-2} \cdot day^{-1}$) that would create anoxic conditions in sediments (Hargrave 1994). When flux rates are low, deposited organic material is rapidly decomposed and remineralized, resulting in minimal accumulation of reduced metabolic by-products. For example, Kelly and Nixon (1984) observed that within two months up to 20% of particulate carbon and 30% of nitrogen filtered from a laboratory seawater system and experimentally deposited in microcosms were released from sediments into overlying water as dissolved inorganic products of decomposition. They speculated that initial rates of decomposition of freshly settled organic matter collected in sediment traps may be even more rapid (5-10%$\cdot day^{-1}$). Such rapid loss reduces the amount of organic matter available for storage in deeper sediment layers. However, the dissolved substances released from sediments are available for widespread dispersion by water mixing and exchange.

Cranston (1994) suggested that organic matter fluxes to sediments could be estimated from carbon burial (CB). He proposed this as an integrative measure of enrichment due to finfish aquaculture, since accumulation is measured after labile organic matter is lost. Several studies have shown that preservation rates of non-labile carbon increase with CB (Müller and Suess 1979). Cranston (1994) examined the relationship in near-shore areas of SWNB by comparing dissolved ammonium and sulfate profiles in sediment pore water with known CB rates. In a comparison of over 100 coastal to deep ocean sites, highest values (2-10 g $C{\cdot}m^{-2}{\cdot}day^{-1}$) occurred at salmon farm sites in the Letang area (SWNB), where carbon concentrations were >3% of sediment dry weight. By comparison, CB at sites in Halifax Harbour, an inlet on the Atlantic coast of Nova Scotia (NS) that receives urban sewage, varied from 0.2 to 0.7 g $C{\cdot}m^{-2}{\cdot}day^{-1}$.

Although CB is an integrative measure that reflects all sources of particulate input to an inlet system, organic carbon accumulation determined by this measure cannot be attributed to point sources (Cranston 1994). There may also be depocenters where burial rates of fine particles are high, whereas in erosional areas of high bottom stress, rates will be much lower. As with observations of particulate matter collected with sediment traps, CB may be highly site-specific.

Tlusty et al. (2000b,c) used EM3000 multibeam acoustic methods to produce detailed bathymetric and backscatter maps in Bay d'Espoir, NL, and Hughes Clarke (2001) described a similar study in Letang Inlet, SWNB. Multibeam swath bathymetry, used for precise hydrographic mapping of bottom depth in these studies, also provided inlet-wide images to identify depocenters where fine-grained sediments focused from a broad area accumulated. Unlike traditional benthic sampling with cores, grabs or photography, the multibeam acoustic method provides an inlet-wide view of sediment properties. As with all remote sampling, ground-truthing and interpretating are necessary to provide insights into processes that affect sediment dynamics and accumulation over a wide area. With accurate geo-referenced positioning and concurrent measurements of tidal amplitude, depth resolution of ±5 cm is possible using the EM3000 system, allowing changes in bottom topography due to sediment accumulation under and near net-pens to be observed.

Variations in acoustic signals may be used to differentiate areas of hard substrates (erosional) from fine-grained (depositional) sediments. However, backscatter strength may also be used to detect areas where organic matter accumulation is sufficient to create anoxic conditions with gas accumulation and changes in sediment porosity. Hughes Clarke et al. (2002) described two acoustic surveys eight months apart at an active salmon farm site in SWNB. Comparison of cage positions in 1996 and 2000 at the time of the acoustic survey showed that anomalous signatures associated with net-pens could persist for at least four years. The observations illustrated local variability associated with the presence of net-pens with distinct acoustic signatures in sediments detectable immediately under cages. There was minimal horizontal displacement consistent with known low currents in the area. Ground-truthing is currently underway to determine specific sediment physical-chemical properties responsible for the acoustic backscatter response.

Bottom substrate type and sediment deposition patterns in any coastal area reflect local bathymetry, hydrography and the input of particulate matter from a variety of sources. Particulate matter formed by autotrophic production of phytoplankton and macrophytes may be combined with material from cliff erosion and suspended matter from river input.

Shoreline erosion and inputs from the Saint John River are major sources of particulate matter contributing to high sedimentation rates (0.5-2 $cm \cdot y^{-1}$) determined by ^{210}Pb and ^{137}Cs geochronology in sediment cores from offshore areas of SWNB (Smith et al. 2002). Lower sedimentation rates (0.2-0.6 $cm \cdot y^{-1}$) occur in inshore areas where aquaculture is concentrated, indicating that deposition in these areas is dominated by local sources. The observations show the importance of evaluating sedimentation rates over a relatively large region in order to differentiate variable sources and rates of accumulation of deposited material.

FOOD WEB STRUCTURE AND FUNCTION

EFFECTS ON WILD POPULATIONS

There is a growing literature on effects of salmon aquaculture on wild stocks of other salmonid and non-salmonid species dealt with in other papers in this series. For example, historic herring migration routes into macrotidal inlets in SWNB may have been altered by the increasing abundance of salmon net-pens. Anecdotal evidence is that herring weirs close to farm sites have reduced catches in comparison with historic landings (Stephenson 1990). The changes, however, must be evaluated in light of widespread reductions in herring populations. Further studies are required to determine if herring population size on traditional bottom spawning grounds has been altered as well as feeding patterns disrupted. With reduced predation, standing stocks of zooplankton normally consumed by herring would conceivably remain higher than in the past.

Alteration of bottom type to more fine-grained sediments through enhanced deposition of flocculated, fine-grained material described above may also account for the speculation that a population of lobsters was displaced from their historic spawning ground after a salmon farm was located at the site (Lawton and Robichaud 1991). However, an opposite effect of salmon farm operations causing aggregations of lobster may also occur. Bottom video tape records from a benthic survey at a finfish farm site in Woods Harbour, NS showed large numbers (up to 3-5 individuals$\cdot 10m^{-2}$) of lobsters (*Homarus americanus*) under net-pens when normal densities would have been much lower (B. Hargrave, unpublished data). Fish feed pellets observed on the bottom probably provided a food supply that attracted lobsters from a larger area. It is also possible that a refuge from harvesting was created within the farm site, since lobster pots are not usually set within a farm lease area. If lobsters are attracted to farm sites, they would be vulnerable to chemical therapeutants and antimicrobial agents used to control parasites and diseases in cultured fish.

A related far-field population effect involves the potential for farmed fish to prey on juveniles of other species (salmon and herring) when they enter cages. This may occur especially at night when lights, used to accelerate farmed salmon growth, attract juveniles of various fish species (C. Levings, West Vancouver Laboratory, Vancouver, BC, personal communication). The consumption of wild feed by caged salmon would have the greatest impact on natural populations where seaward migration routes of juvenile wild species pass through areas occupied by salmon farms. Results from a short-term study using seine nets around net-pens in coastal BC did not show that farmed fish prey on juveniles of other species (Haegele et al. 1991; Black et al. 1992).

There is also the related issue of behavioral changes in vertebrate (mammals and fish) and invertebrate distribution caused by acoustic deterrent and harassment devices used principally to repel seals from salmon farm sites. The topic has a high public profile. During the next few years, studies on effects of seismic exploration on coastal fisheries will be required for environmental impact assessment of offshore hydrocarbon exploration activity. A workshop to develop methods to study these effects was held in Halifax, NS in September 2000 (Thomsen et al. 2001).

The decline in occurrence of killer whales (*Orcinus orca*) and the Pacific white-sided dolphin (*Lagenorhyncus obliquidens*) in the Broughton Archipelago between 1985 and 1998 has been attributed to noise avoidance at distances up to 3.5 km from sea pens (Morton and Symonds 2002). Similar avoidance behavior by harbor porpoise (*Phocoena phocoena*) is thought to have occurred in the SWNB area (Johnston and Woodley 1998). By-catch of harbor porpoise by gill nets can be reduced using acoustic devices (Trippel et al. 1999). American shad (*Alosa sapidissima*) and herring (*Clupea harengus*) are sensitive to high frequencies and sound pressures potentially created by acoustic devices within a few kilometer radius of farm sites (Enger 1967; Mann et al. 1997). Few similar studies exist for invertebrates, although mechanoreceptors are widely distributed in crustacea. Shrimp produce sounds and are sensitive to water movement and vibration (Heinisch and Wiese 1987; Berk 1998). It is therefore possible that these species may also be negatively impacted by seismic activity during hydrocarbon exploration.

CHANGES IN PELAGIC FOOD WEBS

In contrast to numerous studies of localized benthic effects of finfish aquaculture at farm sites discussed below, there have been very few observations of effects on plankton communities (Burd 1997). Stirling and Dey (1990) observed the impact of intensive net-pen fish culture in a Scottish freshwater loch with associated changes in phytoplankton and periphyton. Stratification and restricted circulation of these types of water bodies results in effects of nutrient loading similar to those observed in lakes. One experimental study in 200-L mesocosms over two months simulated seawater slowly flowing over sedimented feed pellets (Parsons et al. 1990). The main effects were that zooplankton were eliminated when oxygen was dramatically reduced (<10% saturation) in bottom water during the first ten days of the experiment. In the absence of grazing pressure, phytoplankton and bacterial cell numbers, phytoplankton production and bacterial heterotrophic activity increased. The results, while produced experimentally, might be relevant to fjord-type inlets with sills or bathymetry that lead to restricted water circulation. Reductions in zooplankton standing stock with oxygen depletion could allow standing stocks of phytoplankton to increase. With sufficient nutrient supplies, higher rates of primary production and increased sedimentation would result in even further oxygen depletion in deep water.

EFFECTS ON INTERTIDAL AREAS

Intertidal areas, subject to daily movements of water and sediment, are locations influenced by broad-scale processes affecting chemical fluxes of mass and dissolved material throughout an inlet system. However, few studies have unequivocally linked the establishment of aquaculture farm sites to environmental or ecological changes in intertidal areas. For example, increased macroalgal abundance that is often observed within intertidal zones may reflect cumulative changes due to combined effects of recruitment, herbivory and nutrient enrichment (Lotze et al. 2000; Worm and Lotze 2000; Worm et al. 2000). Eutrophication can lead to an increased occurrence of macroalgal mats, with significant changes in the structure of intertidal benthic communities (Soulsby et al. 1982; Raffaelli et al. 1998; Thiel et al. 1998). As with naturally occurring algal blooms, it is often difficult to attribute the effects directly to only one factor such as aquaculture development.

One difficulty in observing effects of aquaculture in intertidal areas is that few studies have specifically addressed long-term changes in this zone that might be attributable to the establishment of fish farm sites (Wong et al. 1999; Auffrey and Robinson 2001). Variables such as sediment grain size (texture), nutrients and organic matter are highly variable in intertidal areas, and it is difficult to relate changes to single factors. A common feature of nutrient enrichment in the intertidal zone, however, is an increase in biomass of green macroalgal mats that form a dense cover over the sediment surface (Soulsby et al. 1982). *Enteromorpha* sp. and *Ulva* sp. are the most common mat-forming species (Kolbe et al. 1995). Since nutrients are generally considered to limit macroalgal growth, new sources of dissolved nutrients are likely to be rapidly removed from the water column and incorporated into macroalgal biomass (Chopin and Yarish 1999; Chopin et al. 2000). Campbell (2001) observed a shift in biomass from perennial macrophytes (kelps and seagrasses) to fast-growing macroalgae in a shallow Australian embayment receiving nutrients from sewage and runoff. He concluded that elevated nutrients, particularly ammonium-nitrogen, may allow macroalgae to colonize large near-shore areas where nutrient limitation had previously restricted their distribution.

The proliferation of *Enteromorpha* mats in excess of 30% sediment cover has been observed at selected intertidal sites in SWNB in macrotidal inlets adjacent to salmon farms (Auffrey and Robinson 2001; Auffrey et al. 2002). The observations of changes in numbers and size of soft shell clams (*Mya arenaria*) in relation to algal mat cover are consistent with other studies that show negative effects on mollusc growth rates attributable to the creation of anoxic conditions within and below macroalgal mats. Since there is a relationship between body and gonad size, reduced clam biomass will result in decreased egg production and recruitment. However, harvesting statistics for the soft shell clam fishery in the Bay of Fundy show that abundance has fluctuated throughout its 100-year history (Robinson 1996). Changes in landings reflect many factors such as closures to harvesting areas due to fecal coliform bacteria and, during the 1950s, new predation pressure due to the northward movement of the green crab.

Nutrient enrichment in intertidal zones may affect benthic fauna as well as macroalgae through changes in the rates and nature of deposition of particulate organic matter. The effects may be difficult to quantify, particularly in sloped rocky shorelines where steep physical gradients occur (Bourget et al. 1994). Field studies of epiphytic invertebrates associated with the macrophyte *Ascophyllum nodosum* did not show any significant trends in changes in community composition at various distances up to 1.5 km from salmon farms on the west coast of Scotland, where an organophosphate had been applied to treat ectoparasites (Murison et al. 1997).

A potential far-field impact of finfish aquaculture on intertidal areas not reflected in published literature involves effects of net cleaning to remove biofouling organisms. Communities of attached marine plants and invertebrates that colonize nets may reach considerable biomass. In SWNB, fouling communities with a wet weight of up to 3.5 $kg \cdot m^{-2}$ have been measured (B. Hargrave, unpublished data). One alternative to removal by high-pressure washing is to tow fouled nets into intertidal areas at high tide. During ebb tide, the same washing procedure is carried out with nets stretched horizontally over the intertidal zone. Dislodged plant and animal debris is initially transferred to the intertidal area rather than subtidal sediments. With subsequent tidal cover, however, some fraction of the material is resuspended and re-enters the water column, where it is redistributed to higher intertidal levels and deeper sublittoral areas. The effect parallels natural processes for redistribution of stranded seaweed (wrack) along a shoreline. However, in this case there is a high proportion of animal tissue mixed with plant debris. For this reason, it can become a foraging area for scavenging birds.

CHANGES IN BENTHIC FOOD WEBS

Benthic Metabolism

Meyer-Reil and Köster (2000) have summarized critical points through which sequential changes associated with eutrophication in coastal waters can be identified. Evidence for progressive stages of enrichment includes increases in inorganic and organic nutrients, microbial biomass and enzymatic decomposition of substrates, nitrification, denitrification and benthic nutrient fluxes. Evidence is also accumulating to show that with increasing eutrophication the ratio of autotrophic to heterotrophic microbial processes is reduced with progressively more organic matter respired in sediments than in the water column (Rizzo et al. 1996).

In oligotrophic and mesotrophic coastal marine systems where high turbidity does not limit phytoplankton production, material flow and cycling predominantly occur in the water column. For example, almost two-thirds of total annual oxygen consumption in Chesapeake Bay occurred in the water column (Kemp et al. 1992). In eutrophic, nutrient-rich areas, heterotrophy predominates based largely on stored organic matter in sediments. In coastal areas, this fundamental shift in ecosystem structure may be reflected seasonally: for example, during spring and late summer following input of organic matter sedimented from algal blooms. A similar shift in the balance between pelagic and benthic respiration could occur on an inlet-wide scale in coastal areas as a result of finfish aquaculture activity, if increases in sedimentation of fine-grained particles and associated organic matter are sufficient to cause sulfide accumulation in sediments.

Under hypertrophic conditions, with very high rates of organic sedimentation and restricted oxygen supply, anaerobic processes such as sulfate reduction lead to the build-up of $S^{=}$ and H_2S gas within sediments. Field observations and models of the balance between dissolved $S^{=}$ production, accumulation and sediment-water exchange have shown that processes other than molecular diffusion affect transport into the water column. Gas formed by methane production in deeper sediment layers can be released as bubbles (Roden and Tuttle 1992). The effect of sulfide diffusion from sediments can result in the formation of oxygen depleted bottom water: a problem first observed in stratified fjords with sills in Norway and Scotland (Aure and Stigebrandt 1990). When the toxic gas is released into the water column, as occurs immediately under net-pens with excessively high rates of sedimentation, fish will be highly stressed and may die (Burd 1997).

The transition from predominantly aerobic to anaerobic metabolism along a benthic enrichment gradient can be measured by sulfide accumulation and by incubating undisturbed sediment cores and following changes in dissolved O_2, CO_2 or dissolved nutrients in supernatant water (Hargrave et al. 1993, 1997; Findlay and Watling 1997). Benthic metabolism at net-pen and reference sites >500 m away in SWNB changed from predominantly aerobic to anaerobic respiration at a threshold sulfide concentration between 200 and 300 μM $S^{=}$. Highest levels of $S^{=}$, gas exchange and NH_4^+ release occurred at farm sites that had experienced high rates of organic loading. While some reference sites had slightly elevated rates of benthic respiration and ammonium flux, showing a response to organic enrichment at distances up to 500 m away from cage sites, most far-field samples show lower benthic fluxes characteristic of other coastal areas in the Bay of Fundy.

Microbial Antibiotic Resistance

Another far-field effect of local sources of organic matter produced by finfish farm sites involves the use of therapeutants. This topic is considered by Burridge (this Volume), but it is important to point out here that antibiotics in medicated fish feed have the potential to induce drug resistance in natural microbial populations on an inlet-wide scale. For example, when the commonly used antibiotic oxytetracycline (OTC) was added to marine sediments, concentrations largely disappeared within a few weeks, but the traces of the antibiotic were detectable for up to 18 months (Samuelsen et al. 1992). More significantly, at the end of the initial 10-day treatment, the percentage of resistant bacteria (ratio of numbers growing on substrate ±OTC) was >100% for all sediments tested. Torsvik et al. (1988) and Björklund et al. (1990) also observed that OTC was detectable in fish farm sediments in Norway and Finland for periods of more than one year, with high levels of bacterial resistance measured in both sediment bacteria and isolates from intestines of wild fish. Ervik et al. (1994b) observed antibiotics in fish and wild mussel near a farm site after medicated feed had been administered.

Resistance to OTC has been detected in aerobic bacteria cultured from water, pelletized feed and fingerlings from freshwater Atlantic salmon farms in Chile (Miranda and Zemelman 2002). High levels of OTC resistance (90% minimum inhibitory concentrations (MIC) up to 2000 μg OTC·mL^{-1}) in selected strains suggested that farms may be reservoirs for bacteria with high-level tetracycline resistance. Similar observations of OTC resistance, but with lower MIC values (up to 160 μg OTC·mL^{-1}) were observed in surface sediments under pens and up to 100 m away from salmon farm sites in SWNB (Friars and Armstrong 2002). Pure cultures of *Aeromonas salmonicida* and mixed bacterial populations present in sediments from intertidal sites and embayments where salmon aquaculture has not been carried out showed no resistance to OTC.

Infectious micro-organisms are found in fallowed sediments at abandoned farm sites (Husevag et al. 1991) indicating that, irrespective of antibiotic use, once a disease outbreak has occurred in an area, the probability of re-infection is increased. As might be expected from increased sedimentation of feed pellets and fish wastes near farm sites, observations in Puget Sound showed that the highest numbers of bacteria (as colony forming units) in sediments generally occurred at farm sites (Herwig et al. 1997). However, the proportion of bacteria resistant to OTC declined exponentially with increasing distance from a farm. Levels of increased antibiotic resistance in sediments induced during a brief (10-day) exposure to OTC were detected up to 75 m from the edge of a cage array in Galway Bay (Kerry et al. 1994). After therapy ended, the frequency of resistance decreased exponentially, and within 73 days under-cage samples were not significantly different from background levels. Samuelsen (1989) showed that persistence depended on sedimentation rates at the site after medication. The half-life of OTC under a 4-cm layer of sediment was doubled from 72 to 135 days.

The development of antibiotic resistance may have the potential for human health risk since positive correlations have been reported between antibiotic use and the isolation of drug-resistant bacteria in fish consumed as food (Hastings and McKay 1987; Alderman and Hastings 1998). Son et al. (1997) reported the successful transfer of antibiotic resistance among strains of *Aeromonas hydrophilia* isolated from cultured *Telapia mossambica* via exchange of plasmids, illustrating the potential for the spread of drug resistance in cultured fish. Further studies are clearly required to determine the extent of far-field effects and ecological and biological impacts of antibiotic resistance induced in microbial and other wild populations in areas of intensive finfish aquaculture.

Benthic Macrofauna Communities

There is an extensive literature documenting changes in benthic infauna community structure associated with high levels of nutrient and organic matter additions and resulting stress due to oxygen deficiency (Pearson and Rosenberg 1978; Warwick 1986, 1987; Nilsson et al. 1991; Engle et al. 1994; Diaz and Rosenberg 1995; Burd 1997; Pohle and Frost 1997; Nilsson and Rosenberg 2000). Moderate increases in organic matter supply may stimulate macrofauna production and increase species diversity. However, with increasingly higher rates of organic input, most studies have shown that the local extent of altered benthic community structure and biomass is often limited to distances of less than a few hundred meters (Sowles et al. 1994; Findlay et al. 1995; Brooks 2001). The distance is determined by water depth and current velocity among other factors (Weston 1990; Pohle et al. 1994; Silvert 1994e; Henderson and Ross 1995; Burd 1997; Pohle and Frost 1997) and is therefore different at different farm sites.

Widespread changes in species community composition of benthic macrofauna at spatial scales distant from farm sites are more difficult to detect and have been less studied. There have been numerous studies of macrofauna distribution and production in areas off NS and NB, including sites in SWNB and the Bay of Fundy where finfish aquaculture has developed (Wildish and Peer 1983; Wildish 1985; Wildish et al 1986b; Pocklington et al. 1994; Stewart et al. 2001). These data provide background information on macrobenthic community species composition, biomass and estimated production that can be compared with observations in specific inlet systems where finfish aquaculture has been established.

Only fauna tolerant of low oxygen conditions are able to survive under conditions of high organic sedimentation (Diaz and Rosenberg 1995; Hargrave et al. 1997; Nilsson and Rosenberg 2000; Brooks 2001). However, some benthic infauna taxa are sulfide-tolerant (e.g. some nematode species and the polychaete *Capitella capitata*) (Hargrave et al. 1993; Pocklington et al. 1994; Duplisea and Hargrave 1996). The presence/absence of these 'indicator' species or faunal groups under and around finfish aquaculture sites may show transitions from low (background) levels of organic matter supply to high deposition rates caused by unconsumed feed pellets and fish feces in areas subject to low transport (Weston 1990; Pocklington et al. 1994; Burd 1997; Brooks 2001).

Temporal and spatial scales of changes in benthic macrofauna abundance and biomass have been measured as part of a long-term monitoring program over the past decade near net-pen sites and at more distant locations in SWNB to determine if organic enrichment effects from aquaculture can be detected. Lim and Gratto (1992), Pohle et al. (1994, 2001) and Pohle and Frost (1997) used multivariate, distributional and univariate benthic community analyses to show that measurable bay-wide changes have occurred in benthic indicator macrofauna species and levels of organic matter in sediments between 1989/91 and 1998/99. Initial observations near newly established farm sites indicated that organic enrichment effects were localized to within 30 m of cages. However, after approximately five years, changes were measurable over greater (>200 m) distances. Macrofauna community diversity was most reduced close to a farm site that had been in operation for 12 years, but significant declines in diversity also occurred throughout the inlet system. Two bivalve species (*Nucula delphinodonta* and *N. proxima*) were particularly sensitive indicators of organic enrichment showing initial increases in abundance followed by dramatic decreases at sites with high rates of organic sedimentation.

In a study of similar design using spatial observations, Wong et al. (1999) observed benthic epifauna and infauna at two intertidal locations in SWNB at varying distances from salmon aquaculture sites. Due to the complexity of interdependent abiotic and biotic factors that affect species distribution and composition in intertidal communities, stratified random sampling was required to test for significant differences. While numbers and diversity of epifauna were unrelated to distance from salmon cages, diversity of infauna was significantly higher away (>500 m) than near (<500 m) farm sites. Loss of diversity at distances <500 m may indicate that infauna are more sensitive (less tolerant) to organic matter additions than epifauna (Warwick 1986, 1987), possibly reflecting changes in sediment physical structure (grain size), oxygen supply and sulfide accumulation associated with increased organic matter supply.

Changes in sublittoral and intertidal macro-infauna community composition are consistent with observations associated with finfish cage sites where excessive organic matter sedimentation occurs and leads to the formation of increasingly greater anoxia (Weston 1990; Gowen et al. 1994; Pocklington et al. 1994; Diaz and Rosenberg 1995; Doucette 1995; Hargrave et al. 1997; Wildish et al. 1999; Brooks 2001). There is an extensive body of literature describing changes in sediment geochemical variables (e.g. organic and elemental composition, sulfate reduction and sulfide accumulation) associated with the formation of anoxic sediments. The degree of impact is related to rates of organic matter supply (Hall et al. 1990; Holmer and Kristensen 1992; Hargrave et al. 1993, 1997; Hargrave 1994; Doucette 1995; Holmer and Kristensen 1996). As described above, changes are most readily observed immediately under and adjacent to net-pens, but in some cases the spatial scales of effects may be more than local and exceed 500 m (Hargrave et al. 1993).

Evidence for near-field benthic enrichment effects will be discussed in detail elsewhere in this series, but it is important to emphasize here that under certain circumstances locally anoxic sublittoral sediments may have the potential to cause inlet-wide effects. Since hypoxia is a critical factor that can determine taxonomic structure in marine benthic communities (Grizzle and Penniman 1991; Rosenberg et al. 1992; Diaz and Rosenberg 1995; Nilsson and Rosenberg 1997, 2000), sediments receiving increasing amounts of organic matter within an inlet system could result in an overall reduction in dissolved oxygen. Benthic crustaceans are particularly sensitive to hypoxia (Brante and Hughes 2001), and thus the absence of these taxa may be the first indication of significant alterations in benthic macrofauna community composition due to low oxygen stress.

An example of how local anoxic conditions can have an inlet-wide effect was provided by Ueda et al. (2000), who described the negative effects of seasonal movements of oxygen deficient waters over an intertidal zone on macrobenthos in Dokai Bay, Japan. Urban wastewater effluents created oxygen deficits in mid- and deep-water areas of the bay that were periodically upwelled onto the intertidal zone with dramatic effects of mortality of benthic fauna. Only a few polychaete species survived brief intervals of hypoxic conditions. In this hypertrophic environment, very high rates of organic loading created conditions of oxygen depletion such that bottom water in certain parts of Dokai Bay in late summer had an oxygen concentration of $<0.8\ mg \cdot L^{-1}$. Urbanized inlets without sills and open to exchange with offshore coastal water would not usually be expected to develop such dramatic oxygen deficits.

Observations of conditions of oxygen supply to the sediment-water interface using photographic images (Rhoads and Germano 1982; O'Connor et al. 1989; Grizzle and

Penniman 1991; Valente et al. 1992) were made in SWNB (Letang Inlet) in 1985 (Wildish et al. 1986a). Anoxic conditions caused by discharge of pulp mill effluent had not moved seaward, but depositional and reducing conditions (measured as low redox (Eh) potentials) were observed locally under cages at an aquaculture site. Similar studies with measurements of Eh, total sedimentary sulfide and organic matter have been conducted in Letang Inlet over the past several years (Hargrave et al. 1993, 1995, 1997; Wildish et al. 1999, 2001a,b) as well as recently in BC (Brooks 2001). The results show that Eh and total $S^{=}$ can be used to scale organic matter loading to sediments and associated changes in benthic community structure under and around salmon farm sites. Changes in these variables are spatially and temporally correlated with changes in benthic community species composition and biomass (Brooks 2001; Wildish et al. 2001b) and hence provide measures potentially useful for regulatory purposes. Performance-based standards for salmon aquaculture using thresholds of these indicators for sediment organic enrichment are being developed and applied as part of environmental management guidelines for salmon aquaculture in NB (DELG 2000) and BC (Erickson et al. 2001).

KNOWLEDGE GAPS

1. There is a need to determine sustainable levels of salmon production within coastal regions, inlets or embayments where marine finfish aquaculture is currently practiced in Canada. DFO has statutory obligations under the *Fisheries Act* for fish habitat protection. Research is needed to establish optimal siting criteria to minimize harmful alteration, destruction or disruption of habitat (HADD), to identify sensitive variables for quantifying HADD and to develop methods for cost-effective monitoring.

- Is there an optimal upper limit to stocking density (e.g. 18 $kg \cdot m^{-3}$) for finfish aquaculture sites that is sufficient to avoid problems associated with disease, eutrophication and the onset of oxygen depletion in the water column and formation of anoxic sediments?
- What are the most important factors determining the upper limit to stocking density?
- How can accurate information on stock biomass, rates of food application and yield (feed conversion efficiency) be obtained from industry to construct ecosystem-level models of cumulative environmental impacts of aquaculture development on a broad (inlet-wide) scale? Should this be a condition of license approval?
- How can estimates of carrying capacity be made to reflect cumulative loading from all farm sites in one hydrographic area?

2. Mass balance models of nutrient loading (inorganic and organic) from all sources (natural and anthropogenic) can be used to assess potential additions from finfish aquaculture. Budgets must take into account internal nutrient recycling as well as external sources. Physical and biological processes must be included in biogeochemical models to describe sources, sinks and fluxes of nutrients important for growth of phytoplankton and macrophytes.

- Does the release of particulate and dissolved material significantly enhance natural levels of the same substances in an inlet?
- What are the relative contributions of nutrient addition and BOD due to finfish aquaculture to all other water column and benthic fluxes in an inlet system?
- Do waste product releases from aquaculture and other sources of enrichment alter naturally occurring ratios as well as absolute levels of essential dissolved nutrients?

3. General circulation models must be developed and improved to resolve combined effects of tidal and wind-driven forcing and that reflect complex topography and intertidal drying zones. For mixing and exchange processes to be adequately described, models should be developed to include effects of tidal properties, residual circulation, wind forcing and effects of freshwater input.

- What observations are required to predict and ground-truth particle dispersion and water residence times in areas subject to aquaculture development?
- Can models accommodate appropriate temporal and spatial scales for different questions?
- To what degree can numerical and finite element circulation models be generalized to be useful for predicting particle dispersion, accumulation and burial rates?
- Can numerical tidally driven hydrodynamic three-dimensional finite element models be used to predict water mixing, particle transport and properties, such as dissolved nutrient distributions and BOD, in areas where finfish aquaculture has developed?
- Can these models include interactions with suspended particles by applying particle-related parameters, such as decay coefficients, settling rates and planktonic food web interaction terms?
- Can fully coupled physical-biological models be developed that describe the dynamics and trace pathways of soluble and particulate matter within a coastal system for which these models have been verified?

4. New methods are needed to quantify resuspension processes. Resuspension may redistribute fine material produced locally by finfish aquaculture sites over large areas. An understanding is required of effects of mixing and exchange on the dynamics of particulate matter transport and dispersal. Pathways and rates of transport and the ultimate fate of particulate matter produced at farm sites are determined by these processes.

- Can high resolution (multibeam) seabed mapping be used to provide inlet-wide bottom type classification and identification of depocenters where fine-grained particulate matter accumulates?
- What other rapid survey methods (e.g. videography and sediment image profiling) exist that might be used to identify sediment-water interface processes that affect particle resuspension and post-depositional transport?
- What chemical tracers exist that are useful for tracking dispersion of particulate matter from point source additions at farm sites?

5. New methods and knowledge are required to describe processes, such as flocculation and aggregation, that affect particle dispersion. Particle properties such as grain size, density, concentration and organic coatings affect local dispersion of particulate matter around finfish farm sites and determine the extent of far-field transport.

- What laboratory and field observations are required to determine the role of these processes in the dynamics of suspended matter on a coastal (inlet-wide) scale?
- What are the temporal and spatial scales required to observe long-term trends in changes in patterns of sedimentation?
- A major gap is knowledge of the behavior of fish feed and wastes in suspension and on the bottom (e.g. settling rates and erosion thresholds).

6. Studies are required to determine if the frequency and location of HABs or plankton blooms are related to the expansion of finfish aquaculture. Enhanced nutrient inputs from farm sites may alter algal species composition and biomass. Harmful algal species, naturally present in low abundance, may be stimulated to form blooms that are harmful or toxic to caged fish. Natural variations in dissolved oxygen may be also enhanced during bloom conditions resulting in respiratory stress for cultured fish.

- Can the frequency and location of HABs or plankton blooms in general be related to the historic expansion of the finfish aquaculture industry in a given area?
- What methods exist or can be developed to predict the timing and extent of HABs or shifts in species due to alterations in dissolved nutrient concentrations and ratios?
- Do HABs have potentially negative effects on other naturally occurring species?

7. New studies are required to determine what changes occur in water column variables in areas of intensive finfish aquaculture. In comparison to benthic studies, there have been very few investigations of planktonic communities in areas of intensive finfish aquaculture. Low oxygen conditions around net-pens could cause behavioral changes in planktonic species capable of an avoidance response. Alternatively, increased availability of suspended matter released from cages could attract planktonic species.

- Do changes in the organic composition and nature of suspended matter that may be associated with finfish aquaculture alter food supplies to filter feeding planktonic organisms?
- Are there predictable interactions between zooplankton standing stocks, water/sediment oxygen depletion and phytoplankton productivity that can be used to link processes in the pelagic food web with rates of sedimentation and oxygen depletion in deep water?
- Do farmed salmon prey on juveniles of other species that enter net-pens?
- Does the ratio of heterotrophic respiration shift from a pelagic to benthic-dominated system with increasing organic enrichment?

8. Few studies have documented environmental or ecological changes in intertidal areas that can be unequivocally linked to the establishment of aquaculture sites. Intertidal

areas may be either sources or sinks for suspended material near finfish farm sites depending on local hydrography.

- What variables exist to demonstrate cause-effect relationships in changes in sedimentological and biological variables in intertidal areas that can be related to finfish aquaculture development?
- What is the relative impact of increases in dissolved nutrient availability and suspended particulate matter sedimentation to biological communities in the intertidal zone in comparison to anticipated inputs from finfish aquaculture?
- Are the natural distribution, standing stock or species composition of macroalgae in intertidal and subtidal areas altered at locations proximate to finfish aquaculture sites?

9. Mass balance and numerical models are required to link production and external loading with aerobic and anaerobic oxidation of organic matter (pelagic and benthic), sedimentation and sulfide accumulation in sediment. If advection and diffusion of dissolved oxygen to the sediment surface are insufficient to maintain oxic conditions, reduced metabolic products of bacterial decomposition of organic matter accumulate within surface layer sediments. In extreme cases of anoxia, mats of white sulfur bacteria (*Beggiatoa spp.*) form on the sediment surface at the oxic-anoxic interface.

- How can circulation and dispersion models be coupled to predicted rates of organic matter consumption in the water column and sediments?
- What methods exist for the determination of the assimilative capacity of biota in the water column and sediments for additional organic matter loading?

10. Studies are required to determine the extent of far-field effects of antibiotics added to fish feed pellets on microbial and other wild populations in areas of intensive fish farming. Antibiotics in fish feed pellets are inefficiently assimilated by cultured finfish and their uncontained release into aquatic systems can promote the selective growth of antibiotic resistant bacterial strains.

- Do commonly used antibiotics persist in sediments and, if so, how far from sites of application?
- Is treatment dose or persistence associated with the induction of antibiotic resistance?
- Are antibiotic-resistant bacteria strains transferred from sediments to wild fish and invertebrates?

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ECOSYSTEM LEVEL EFFECTS OF MARINE BIVALVE AQUACULTURE

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EXECUTIVE SUMMARY

This paper reviews the present state of knowledge on environmental issues related to bivalve aquaculture, with emphasis on suspended mussel culture. Material reviewed includes Canadian and international studies on the role of wild and cultured bivalve populations in controlling ecosystem-level dynamics. The focus is on identifying potential changes in ecosystem processes (material and energy fluxes, and nutrient cycling) at the coastal ecosystem scale. Potential mechanisms for ecosystem-level effects include the utilization of particulate food resources by cultured bivalves and associated fauna, the subsequent release of unutilized materials in dissolved (urine) and particulate (feces and pseudofeces) form, and the removal of minerals from the system in the bivalve harvest. The potential consequences to coastal ecosystems from intensive bivalve aquaculture are summarized in the following sections.

CONTROL OF SUSPENDED PARTICLE DYNAMICS

Dense bivalve populations may exert a strong influence on suspended particulate matter (including phytoplankton, detritus, and some auto- and heterotrophic picoplankton and micro-zooplankton) in some coastal systems through their huge capacity to clear particles from the surrounding water (Dame 1996). Grant (2000) studied 15 embayments in Prince Edward Island (PEI) and concluded that the mussel biomass under culture in 12 of these embayments was potentially capable of removing food particles much faster than tidal exchange could replace them. Similarly, Meeuwig et al. (1998) concluded that mussel culture operations in many PEI embayments significantly reduce phytoplankton biomass through grazing. Similar conclusions have been reached for numerous international coastal regions (reviewed by Dame 1996). These relatively simple calculations of the time required for bivalve populations to clear the water column of particles indicate that intensive bivalve aquaculture has the capacity to alter matter and energy flow at the coastal ecosystem scale. However, gaps in knowledge exist regarding a number of important processes that could potentially mitigate the suspected impact of bivalve feeding. These include the following: (1) the effects of physical processes such as water column stratification, mixing and flow velocity, and spatially dependent tidal flushing; (2) the replenishment of food particles through primary production within the embayment; (3) bivalve-mediated optimization of primary production (Prins et al. 1995);

and (4) the large flexibility of bivalve feeding responses to environmental variations (Cranford and Hill 1999).

A strong indication that bivalve filter-feeders are able to control suspended particulate matter in some coastal systems comes from documented ecosystem changes that occurred after large biomass variations in natural and cultured bivalve populations. Population explosions of introduced bivalve species in San Francisco Bay and dramatic reductions in oyster populations in Chesapeake Bay have been implicated as the cause of large changes in phytoplankton biomass and production experienced in these systems (Nichols 1985; Newell 1988; Nichols et al. 1990; Alpine and Cloern 1992; Ulanowicz and Tuttle 1992). Research on the whole-basin environmental effects of bivalve aquaculture in France and Japan indicate that intense bivalve culture in these regions led to changes in particulate food abundance and quality, resulting in large-scale growth reduction and high mortalities in the cultured bivalves (Héral et al. 1986; Aoyama 1989; Héral 1993). Speculation that intense bivalve culture can affect coastal ecosystems by reducing excess phytoplankton associated with eutrophication has been supported by some laboratory and field observations, but has not been rigorously proven.

DIVERSION OF MATERIALS TO BENTHIC FOOD WEBS

The feeding activity of bivalve filter-feeders results in the packaging of fine suspended material into larger feces and pseudofeces that rapidly settle to the seabed, especially under conditions with slow or poor water flushing and exchange. These activities divert primary production and energy flow from planktonic to benthic food webs. While the dynamics of bivalve feces deposition (settling velocity, disaggregation rate and resuspension) are poorly understood, enhanced sedimentation under shellfish culture is well documented. Mortality and fall-off of cultured bivalves, induced by seasonal colonization by fouling organisms, can result in additional acute benthic organic loading.

The spatial scale and degree of seabed organic enrichment effects caused by the increased vertical flux of naturally occurring particles are dependant on the biomass of cultured bivalves, local hydrographic conditions, and the presence of additional organic inputs from other natural and anthropogenic activities. The recycling of organic biodeposits under suspended mussel culture operations in PEI, and at several other international regions, has been shown to have local to inlet-wide benthic impacts (Dahlback and Gunnarsson 1981; Tenore et al. 1982; Mattsson and Linden 1983; Kaspar et al. 1985; Shaw 1998; Stenton-Dozey et al. 1999; Mirto et al. 2000; Chamberlain et al. 2001). The increased oxygen demand in sediments from mussel biodeposits can, under certain conditions, result in the generation of an anaerobic environment that promotes ammonification and sulfate reduction, increased sediment bacterial abundance, and changes in benthic community structure and biomass. Aquaculture is not solely responsible for such impacts in PEI, as many basins are also stressed by nutrient enrichment from agricultural run-off. Observations of seabed impacts under mussel lines in PEI are, therefore, not directly transferable to bivalve culture sites in many other regions of Canada. Biodeposition patterns and the dispersion *of bivalve biodeposits are also controlled by water depth and local water movement. Slight differences in these physical properties can result in marked differences in the degree of impact observed on seabed geochemistry and communities under different suspended mussel culture sites (Chamberlain et al. 2001). Further research is needed to assess the ability of different coastal regions to resist or assimilate the effects of increased organic enrichment through a variety of physical and biogeochemical processes.*

The increased coupling of planktonic and benthic food webs by cultured bivalves has the potential to change energy flow patterns in coastal ecosystems, including altering food availability to zooplankton and larval fish (Horsted et al. 1988; Newell 1988; Doering et al. 1989). Bivalve filter-feeders have a competitive advantage over zooplankton for food resources because they are able to respond immediately to increased food availability, while zooplankton must go through a complete life cycle before being able to fully exploit increased food resources. Direct ingestion of zooplankton by bivalves may also reduce zooplankton abundance (Horsted et al. 1988; Davenport et al. 2000). However, effects of bivalve culture on zooplankton communities are largely speculative owing to the limited research conducted.

ALTERED COASTAL NUTRIENT DYNAMICS

The consumption and deposition of suspended particulate matter by bivalves, as well as the excretion of dissolved nutrients, can play a significant role in controlling the amounts and forms of nitrogen in coastal systems and the rate of nitrogen cycling (reviewed by Dame 1996). This transformation and translocation of matter by bivalves appears to exert a controlling influence on nitrogen concentrations in some coastal regions (Dame et al. 1991) and can provide a means of retaining nutrients in coastal areas, where they are recycled within detrital food chains, rather than being more rapidly exported (Jordan and Valiela 1982). Benthic nutrient mineralization can increase at culture sites as a result of the increased organic matter sedimentation, greatly speeding up the rate of nitrogen cycling (Dahlback and Gunnarsson 1981; Kaspar et al. 1985; Feuillet-Girard et al. 1988; Barranguet et al. 1994; Grant et al. 1995). The high flux of ammonia excreted from dense bivalve populations may have a major effect on phytoplankton production (Maestrini et al. 1986; Dame 1996) and may potentially contribute to more frequent algal blooms, including those of the domoic-acid-producing diatom Pseudo-nitzschia multiseries (Bates 1998; Bates et al. 1998). Aquaculture-induced changes in the relative concentrations of silica, phosphorus and nitrogen (e.g. Hatcher 1994) may also favor the growth of other harmful phytoplankton classes (Smayda 1990), but this has yet to be observed in nature. Bivalve aquaculture may also play a significant role in nutrient cycling in coastal systems, as nutrients stored in the cultured biomass are removed by farmers and the nutrients are no longer available to the marine food web. Kaspar et al. (1985) suggested that the harvesting of cultured mussels may lead to nitrogen depletion and increased nutrient limitation of primary production, but there is little direct evidence of environmental effects. The retention and remineralization of limiting nutrients in coastal systems is necessary to sustain system productivity, but the potential impacts of bivalve cultures on coastal nutrient dynamics is poorly understood.

CUMULATIVE ENVIRONMENTAL EFFECTS

Any attempt to assess ecosystem-level effects of bivalve aquaculture must consider the complexity of natural and human actions in estuarine and coastal systems. Infectious diseases associated with intense bivalve culture, as well as exposure of cultured organisms to 'exotic' pathogens introduced with seed or broodstock, can have a significant and perhaps more permanent impact on ecosystems than the direct impact of the bivalves themselves (Banning 1982; ICES 1995; Bower and McGladdery 1996; Hine 1996; Renault 1996; Minchin 1999; Miyazaki et al. 1999). The presence of additional ecosystem stressors can also influence the capacity of bivalves to impact the ecosystem. The effects of chemical contaminants and habitat degradation are complex, but are well documented as having the potential to adversely affect bivalve health. Bivalve neoplasias show strong correlations to heavily contaminated environments (Elston et al. 1992), and the severity of infection is related to sub-optimal growing conditions (Elston 1989). Dissolved contaminants are frequently scavenged onto particulate matter, a mechanism which increases their availability for wild and cultured filter-feeders to ingest. Weakened bivalves with impeded feeding activity, along with spawning failure or poor quality spawn, can all contribute to morbidity, mortality and fall-off.

Land-use practices that transport sediments into estuaries can impact coastal water quality. Cultured bivalves and their support structures could alter sedimentation patterns within embayments, resulting in accelerated deposition of fine-grained sediment. Presently, there is no consensus on whether dense bivalve populations cause a net increase or decrease in sedimentation rates in coastal regions. However, if bivalve cultures influence the natural equilibrium among the major factors controlling sediment aggregation rate, sedimentary conditions within coastal regions may be altered.

INTEGRATION OF ECOSYSTEM EFFECTS

The available literature has shown that extensive bivalve culture has the potential, under certain conditions, to cause cascading effects through estuarine and coastal foodwebs, altering habitat structure, species composition at various trophic levels, energy flow and nutrient cycling. Simulation modelling has been one of the more focused approaches to assessing the net ecosystem impact of bivalve interactions with ecosystem components. Modelling can quantitatively and objectively integrate the potential negative ecosystem effects of the impact of mussel grazing on phytoplankton, zooplankton and the benthos, with the potentially positive effects of increased recycling of primary production and retention of nutrients in coastal systems (Fréchette and Bacher 1998). For example, such an integrative approach can help to assess whether or not the severity of ecosystem effects in different coastal areas are regulated by water motion and mixing. Numerical models can also be directed toward assessment of system productive capacity, fish/land-use interactions, farm management and ecosystem health. Past work has provided an excellent means of identifying gaps in knowledge.

A variety of methods has been applied to assessing the environmental interactions of bivalve aquaculture operations (Grant et al. 1993; Dowd 1997; Grant and Bacher 1998;

Smaal et al. 1998; Meeuwig 1999), but there is no standard assessment approach. Fully coupled biological-physical models may be envisioned (e.g. Prandle et al. 1996; Dowd 1997) that predict ecosystem changes in chlorophyll, nutrients and other variables of interest as a function of culture density and location. To do this, shellfish ecosystem models must be integrated with information on water circulation, mixing and exchange to account for transport and spatial redistribution of particulate and dissolved matter. Box models (Raillard and Menesguen 1994; Dowd 1997; Chapelle et al. 2000) offer a practical means to couple coastal ecosystem models with physical oceanographic processes. The bulk parameterizations of mixing required for these box models can be derived directly from complex hydrodynamic models (Dowd et al. 2002). Another promising avenue for improving ecosystem models is the use of inverse, or data assimilation, methods (Vallino 2000). These systematically integrate available observations and models, thereby combining empirical and simulation approaches, and improve predictive skill. Simulation models that focus on estimating mussel carrying capacity and related ecosystem impacts provide powerful tools for quantitative descriptions of how food is captured and utilized by mussels, as well as site-specific information on ecosystem variables and processes (Carver and Mallet 1990; Brylinsky and Sephton 1991; Grant 1996).

RESEARCH NEEDS

Few studies have assessed the potential environmental interactions of the bivalve aquaculture industry, and few quantitative measures presently exist to measure the ecosystem-level effects of this industry. Research on the ecosystem-level impacts of bivalve aquaculture is currently at a relatively early stage of development compared with finfish culture and for many other anthropogenic activities. As a result, ecologically relevant studies are needed on many topics, particularly the long-term responses of major ecosystem components (phytoplankton, zooplankton, fish, benthos, as well as the cultured bivalves) to bivalve-induced changes in system energy flow and nutrient cycling. The following general research areas have been identified to address gaps in knowledge:

- Ecological role of bivalve filter-feeders: *accurately quantify the density-dependant role of bivalves in controlling phytoplankton and seston concentrations, including studies of hydrodynamics, bivalve ecophysiology, and phytoplankton community and productivity responses to grazing pressure.*

- Organic loading: *identify important processes controlling the severity of seabed organic enrichment impacts caused by bivalve biodeposits and determine the capacity of different coastal ecosystems to assimilate or recover from the effects of aquaculture-related organic loading.*

- Nutrient dynamics: *develop a predictive understanding of the potential effects of bivalve aquaculture on nutrient concentrations, elemental ratios and rate of cycling in coastal systems, and study the consequences of altered nutrient dynamics to phytoplankton communities and blooms, including harmful algal blooms.*

- Ecosystem structure: *investigate the effects of bivalve culture on ecosystem structure resulting from direct competition between bivalves, zooplankton and epibionts for trophic resources, and the transfer of energy and nutrients to benthic food webs.*

- Numerical modelling: *integrate knowledge obtained on the consequences of bivalve culture on ecosystem structure and function through the use of ecosystem modelling to assess the net impact of aquaculture activities on major system components and to address issues of aquaculture productive capacity and sustainability.*

- Ecosystem status: *develop a scheme for classifying and assessing the state of ecosystem functioning for regions supporting bivalve aquaculture. Integrate multiple ecosystem stressors from anthropogenic land- and marine-use in ecosystem studies of culture systems.*

EFFETS ÉCOSYSTÉMIQUES DE L'ÉLEVAGE DE BIVALVES MARINS

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RÉSUMÉ

Dans cet article, nous passons en revue l'état actuel des connaissances sur les enjeux environnementaux de l'élevage de bivalves, en mettant l'accent sur la culture de moules en suspension. Nous abordons des études menées au Canada et à l'étranger sur le rôle des populations de bivalves sauvages ou de culture dans la dynamique des écosystèmes. Nous visons surtout à déterminer les modifications possibles des processus écologiques (flux de matière et d'énergie et cycles des éléments nutritifs) à l'échelle de l'écosystème côtier. Les mécanismes possibles des effets écosystémiques comprennent l'utilisation de particules de nourriture par les bivalves d'élevage et la faune connexe, l'excrétion subséquente de matières non utilisées sous formes dissoute (urine) ou particulaire (fèces et pseudofèces) et l'enlèvement de minéraux de l'écosystème par la récolte des bivalves. Les répercussions possibles de l'élevage intensif de bivalves sur les écosystèmes côtiers sont résumées dans les sections suivantes.

DYNAMIQUE DES PARTICULES EN SUSPENSION

Dans certains écosystèmes côtiers, les populations denses de bivalves, en raison de leur énorme capacité de filtrer les particules dans l'eau (Dame 1996), peuvent fortement influer sur les particules en suspension (qui comprennent le phytoplancton, le détritus, le picoplancton autotrophe ou hétérotrophe et le microzooplancton). Dans une étude de 15 baies à l'Île-du-Prince-Édouard, Grant (2000) a conclu que la biomasse des moules élevées dans 12 de ces baies pouvait consommer les particules de nourriture beaucoup plus rapidement que ces dernières pouvaient être remplacées par les apports des marées. Dans le même ordre d'idées, Meeuwig et al. (1998) ont montré que l'élevage des moules dans de nombreuses baies de l'Î.-P.-É. réduit considérablement la biomasse du phytoplancton. Des conclusions semblables ont été tirées pour de nombreuses régions côtières du monde (études passées en revue par Dame [1996]). Les calculs relativement simples du temps requis pour que des populations de bivalves filtrent toutes les particules de la colonne d'eau indiquent que l'élevage intensif de bivalves peut modifier les flux de matière et d'énergie à l'échelle de l'écosystème côtier. Toutefois, nos connaissances présentent des lacunes en ce qui a trait à un certain nombre de processus importants qui pourraient atténuer l'impact de l'alimentation des bivalves, notamment : 1) les effets de processus physiques comme la stratification et le mélange de la colonne d'eau, la vitesse

des courants et le renouvellement de l'eau par les marées; 2) le renouvellement des particules de nourriture par la production primaire; (3) l'optimisation de la production primaire par l'intermédiaire des bivalves (Prins et al., 1995); 4) la grande souplesse de la réaction alimentaire des bivalves aux variations environnementales (Cranford et Hill, 1999).

L'observation de modifications de l'écosystème qui ont suivi d'importantes variations de populations naturelles ou cultivées de bivalves indique clairement que les bivalves filtreurs peuvent réguler les particules en suspension dans certains écosystèmes côtiers. On a attribué à des flambées de populations de bivalves introduits dans la baie de San Francisco et à des chutes des populations d'huîtres dans la baie Chesapeake les grandes variations dans la biomasse et la production de phytoplancton observées dans ces écosystèmes (Nichols, 1985; Newell, 1988; Nichols et al., 1990; Alpine et Cloern, 1992; Ulanowicz et Tuttle, 1992). Des recherches concernant les effets environnementaux de l'élevage de bivalves sur des bassins entiers en France et au Japon ont montré que la conchyliculture intensive dans ces régions a modifié l'abondance et la qualité des particules de nourriture, ce qui a entraîné une réduction de la croissance et de fortes mortalités chez les bivalves d'élevage (Héral et al., 1986; Aoyama, 1989; Héral, 1993). Certaines observations faites en laboratoire et sur le terrain appuient l'hypothèse selon laquelle l'élevage intensif de bivalves peut influer sur les écosystèmes côtiers en réduisant le phytoplancton excédentaire associé à l'eutrophisation, mais l'hypothèse n'a pas été rigoureusement vérifiée.

DÉTOURNEMENT DE MATIÈRES VERS LES RÉSEAUX TROPHIQUES BENTHIQUES

L'alimentation des bivalves filtreurs a pour effet d'agglutiner de la matière fine en suspension en des fèces et pseudofèces de plus grande taille qui se déposent rapidement sur le fond marin, surtout dans des conditions de faible renouvellement ou échange d'eau. Cette activité trophique détourne de la production primaire et des flux d'énergie des réseaux trophiques planctoniques vers les réseaux benthiques. La dynamique de sédimentation des fèces de bivalves (vitesse de sédimentation, taux de désagrégation et remise en suspension) est méconnue, mais il est bien établi que la sédimentation est accrue sous les installations conchylicoles. La mortalité et la tombée de bivalves d'élevage, attribuables à la colonisation saisonnière par des salissures, peuvent occasionner d'importants apports supplémentaires de matière organique au milieu benthique.

L'étendue spatiale et le niveau de l'enrichissement du fond marin en matière organique causé par la sédimentation accrue dépendent de la biomasse des bivalves d'élevage, des conditions hydrographiques locales et de la présence d'autres apports organiques naturels ou anthropiques. Il a été montré que le recyclage des dépôts organiques sous des élevages de moules en suspension à l'Î.-P.-É. et ailleurs au monde a des incidences sur le milieu benthique à une échelle spatiale variant de petite à moyenne (Dahlback et Gunnarsson, 1981; Tenore et al., 1982; Mattsson et Linden, 1983; Kaspar et al., 1985; Shaw, 1998; Stenton-Dozey et al., 1999; Mirto et al., 2000; Chamberlain et al., 2001).

Dans certaines conditions, la demande accrue en oxygène des sédiments recevant de la matière organique provenant de mytilicultures peut donner un milieu anaérobie qui favorise l'ammonification, la réduction des sulfates, un accroissement de l'abondance des bactéries dans le sédiment et des modifications de la structure et de la biomasse de la communauté benthique. À l'Î.-P.-É., ces impacts ne sont pas exclusivement attribuables à l'aquaculture, car de nombreux bassins sont également stressés par l'enrichissement en éléments nutritifs provenant du lessivage des terres cultivées. Par conséquent, les observations d'impacts sur le fond marin sous les cordes à moules à l'Î.-P.-É ne sont pas directement applicables aux sites d'élevage de bivalves dans de nombreuses autres régions du Canada. La profondeur de l'eau et les mouvements d'eau à l'échelle locale déterminent aussi le régime de sédimentation organique et la dispersion des dépôts provenant des bivalves. De légères différences dans ces propriétés physiques peuvent produire des différences marquées dans le niveau d'impact observé sur la géochimie et les communautés du fond marin sous des élevages de moules en suspension (Chamberlain et al., 2001). Des études approfondies sont nécessaires pour évaluer la capacité de différentes régions côtières à résister aux effets de l'enrichissement accru en matière organique, ou à assimiler ces apports, par divers processus physiques ou biogéochimiques.

Le couplage accru des réseaux trophiques planctonique et benthique attribuable aux bivalves d'élevage peut modifier le régime de flux d'énergie dans les écosystèmes côtiers, notamment la disponibilité de nourriture pour le zooplancton et les larves de poisson (Horsted et al., 1988; Newell, 1988; Doering et al., 1989). Les bivalve filtreurs possèdent un avantage concurrentiel sur le zooplankton pour l'obtention de ressources alimentaires parce qu'ils peuvent réagir immédiatement à une disponibilité de nourriture accrue, tandis que le zooplancton doit passer un cycle vital entier avant d'être en mesure d'exploiter pleinement les ressources alimentaires accrues. En outre, l'ingestion de zooplancton par les bivalves peut réduire l'abondance du zooplancton (Horsted et al., 1988; Davenport et al., 2000). Toutefois, nos connaissances des effets de l'élevage de bivalves sur les communautés zooplanctoniques reposent largement sur des hypothèses puisque peu de recherche a été effectuée à cet égard.

MODIFICATION DE LA DYNAMIQUE DES ÉLÉMENTS NUTRITIFS EN MILIEU CÔTIER

La consommation de particules en suspension par les bivalves, la sédimentation de la matière qu'ils produisent ainsi que leur excrétion d'éléments nutritifs dissous peuvent jouer un rôle important dans la régulation des quantités et des formes d'azote et du taux de recyclage de cet élément dans les écosystèmes côtiers (synthèse de Dame, 1996). Ces transformation et translocation de matière par les bivalves semblent exercer une influence déterminante sur les concentrations d'azote dans certaines régions côtières (Dame et al., 1991) et peuvent constituer un moyen de retenir des éléments nutritifs dans des zones côtières, où ceux-ci sont recyclés dans des chaînes alimentaires détritiques, plutôt que d'être rapidement exportés (Jordan et Valiela, 1982). La minéralisation benthique des substances nutritives peut augmenter aux sites d'élevage en raison de la sédimentation accrue de matière organique, qui accélère beaucoup la vitesse de *recyclage de l'azote (Dahlback et Gunnarsson, 1981; Kaspar et al., 1985; Feuillet-Girard et al., 1988; Barranguet et al., 1994; Grant et al., 1995). Le flux élevé d'ammoniac excrété par les denses populations de bivalves peut avoir un effet important sur la production phytoplanctonique (Maestrini et al., 1986; Dame, 1996) et pourrait même contribuer à accroître la fréquence des proliférations d'algues, y compris de Pseudo-nitzschia multiseries, une diatomée qui produit de l'acide domoïque (Bates, 1998; Bates et al., 1998). Des changements attribuables à l'aquaculture dans les concentrations relatives de silice, de phosphore et d'azote (p. ex. Hatcher, 1994) peuvent aussi favoriser la croissance d'autres classes de phytoplancton nuisible (Smayda, 1990), mais cela n'a pas encore été observé dans le milieu naturel. L'élevage de bivalves peut également jouer un rôle important dans le cycle des éléments nutritifs dans les écosystèmes côtiers puisque les éléments nutritifs stockés dans la biomasse des bivalves d'élevage récoltés par les aquaculteurs ne sont plus disponibles au réseau trophique marin. Selon Kaspar et al., (1985), la récolte de moules d'élevage peut entraîner l'épuisement de l'azote et accroître la mesure dans laquelle la production primaire est limitée par le manque d'éléments nutritifs, mais il existe peu de preuves directes d'effets sur le milieu. La rétention et la reminéralisation des éléments nutritifs limitants en milieu côtier sont nécessaires au maintien de la productivité de l'écosystème, mais les impacts possibles de l'élevage de bivalves sur la dynamique des éléments nutritifs en milieu côtier sont méconnus.*

EFFETS ENVIRONNEMENTAUX CUMULATIFS

Toute tentative d'évaluation des effets écosystémiques de l'élevage de bivalves doit tenir compte de la complexité des processus naturels et des activités humaines dans les écosystèmes estuariens ou côtiers. Les maladies infectieuses liées à l'élevage intensif de bivalves et l'exposition de ceux-ci à des agents pathogènes « exotiques » introduits avec du naissain ou des géniteurs peuvent avoir un impact important sur les écosystèmes, voire même un impact plus permanent que l'impact direct des bivalves eux-mêmes (Banning, 1982; ICES, 1995; Bower et McGladdery, 1996; Hine, 1996; Renault, 1996; Minchin, 1999; Miyazaki et al., 1999). La présence d'autres facteurs d'agression de l'écosystème peut influer sur la capacité des bivalves de nuire à l'écosystème. Les effets des contaminants chimiques et de la dégradation de l'habitat sont complexes, mais il est bien établi qu'ils peuvent nuire à la santé des bivalves. Les néoplasies chez les bivalves présentent de fortes corrélations avec les milieux très contaminés (Elston et al., 1992), et la gravité de l'infection est liée à des conditions de croissance sous-optimales (Elston, 1989). Les contaminants dissous se fixent fréquemment aux particules, ce qui accroît leur ingestion par les filtreurs sauvages ou d'élevage. Les bivalves affaiblis dont l'alimentation est entravée ainsi que l'échec de la reproduction ou la mauvaise qualité du frai peuvent contribuer à la morbidité, à la mortalité et à la tombée des bivalves.

Les utilisations des terres qui entraînent le transport de sédiments vers les estuaires peuvent nuire à la qualité des eaux côtières. Les bivalves d'élevage et les structures sur lesquelles ils croissent peuvent modifier les régimes de sédimentation dans les échancrures de la côte en accélérant le dépôt de sédiments fins. Il n'existe actuellement aucun consensus à savoir si les populations denses de bivalves produisent une hausse ou *une baisse nette des taux de sédimentation dans les régions côtières. Cependant, si l'élevage de bivalves influe sur l'équilibre naturel entre les principaux facteurs qui déterminent le taux d'agrégation des sédiments, les conditions de sédimentation pourraient être modifiées dans les régions côtières.*

INTÉGRATION DES EFFETS ÉCOSYSTÉMIQUES

La littérature existante montre que, dans certaines conditions, l'élevage intensif de bivalves peut avoir des effets en cascade sur les réseaux trophiques estuariens ou côtiers, en modifiant la structure des habitats, la composition spécifique de divers niveaux trophiques, les flux d'énergie et les cycles des éléments nutritifs. Les modèles de simulation constituent une des méthodes les plus ciblées pour évaluer l'impact net sur l'écosystème des interactions des bivalves avec les composantes de l'écosystème. La modélisation peut intégrer de façon quantitative et objective les effets écosystémiques potentiellement négatifs de l'alimentation des moules sur le phytoplancton, le zooplancton et le benthos et les effets potentiellement positifs de l'accroissement du recyclage de la production primaire et de la rétention des éléments nutritifs dans les écosystèmes côtiers (Fréchette et Bacher, 1998). Par exemple, une démarche intégrative de ce type peut permettre d'évaluer si les mouvements et le mélange de l'eau déterminent ou non la gravité des effets écosystémiques dans différentes régions côtières. Des modèles numériques peuvent aussi servir à évaluer la capacité de production de l'écosystème, les interactions entre l'utilisation des terres et le poisson, la gestion des exploitations aquacoles et la santé de l'écosystème. Les travaux réalisés par le passé constituent un excellent moyen de relever les lacunes dans nos connaissances.

Diverses méthodes ont été utilisées pour évaluer les interactions environnementales des élevages de bivalves (Grant et al., 1993; Dowd 1997; Grant et Bacher, 1998; Smaal et al., 1998; Meeuwig, 1999), mais il n'existe aucune méthode d'évaluation normalisée. On peut envisager des modèles biologiques et physiques entièrement intégrés (p. ex. Prandle et al., 1996; Dowd, 1997) qui permettent de prédire les modifications des concentrations de chlorophylle et d'éléments nutritifs ainsi que d'autres variables d'intérêt en fonction de l'intensité et de l'emplacement des élevages. À cette fin, des données sur la circulation, le mélange et l'échange d'eau doivent être intégrées aux modèles des effets écosystémiques des mollusques pour tenir compte du transport et de la redistribution spatiale des matières particulaires et dissoutes. Des modèles à compartiments (Raillard et Menesguen 1994; Dowd 1997; Chapelle et al., 2000) offrent un moyen pratique de coupler les modèles de l'écosystème côtier avec les processus océaniques physiques. Les paramétrisations globales du mélange requises pour ces modèles à compartiments peuvent être calculées directement à partir de modèles hydrodynamiques complexes (Dowd et al., 2002). L'utilisation de méthodes inverses, ou d'assimilation de données, constitue une autre démarche prometteuse pour améliorer les modèles de l'écosystème (Vallino, 2000). En intégrant systématiquement les observations et modèles disponibles, ces méthodes combinent des démarches empiriques et de simulation et améliorent la capacité de prévision. Les modèles de simulation axés sur l'estimation de la capacité du milieu à soutenir des moules et des impacts écosystémiques connexes constituent des outils puissants pour décrire quantitativement l'obtention et l'utilisation de nourriture *par les moules et fournir de l'information propre à chaque site sur les variables et processus écosystémiques (Carver et Mallet, 1990; Brylinsky et Sephton, 1991; Grant, 1996).*

BESOINS EN RECHERCHES

Peu d'études ont été réalisées pour évaluer les interactions environnementales possibles de l'élevage des bivalves, et il existe peu de mesures quantitatives des effets écosystémiques de cette industrie. La recherche sur les impacts écosystémiques de l'élevage de bivalves est actuellement à un stade de développement peu avancé par rapport à la recherche sur les effets de la pisciculture et de nombreuses autres activités humaines. Il faut donc effectuer des études écologiques pertinentes sur de nombreux sujets, en particulier sur les réactions à long terme des principales composantes de l'écosystème (phytoplancton, zooplancton, poisson et benthos, en plus des bivalves d'élevage) aux modifications, attribuables aux bivalves, des flux d'énergie et des cycles des éléments nutritifs. Les domaines de recherche généraux suivants ont été relevés en vue de combler les lacunes dans nos connaissances :

- Rôle écologique des bivalves filtreurs : *quantifier avec exactitude le rôle de la densité des bivalves dans la régulation des concentrations de phytoplancton et de seston, notamment par des études sur l'hydrodynamique, l'écophysiologie des bivalves et les effets de la pression de broutage sur la composition et la productivité du phytoplancton.*

- Apports organiques : *cerner les processus importants qui déterminent la gravité des impacts d'enrichissement organique du fond marin causés par la sédimentation de matières produites par les bivalves, et déterminer la capacité de différents écosystèmes côtiers à assimiler les apports organiques provenant de l'aquaculture ou à se rétablir des effets de ces apports.*

- Dynamique des éléments nutritifs : *comprendre les effets possibles de l'élevage de bivalves sur les concentrations, les rapports et les taux de recyclage des éléments nutritifs dans les écosystèmes côtiers de façon à pouvoir prédire ces effets, et étudier les répercussions de la modification de la dynamique des éléments nutritifs sur les communautés phytoplanctoniques, notamment en ce qui a trait aux proliférations d'algues nuisibles ou non.*

- Structure de l'écosystème : *étudier les effets de l'élevage de bivalves sur le transfert d'énergie et d'éléments nutritifs aux réseaux trophiques benthiques et sur la structure de l'écosystème qui découle de la compétition alimentaire directe entre les bivalves, le zooplancton et les épibiontes.*

- Modélisation numérique : *intégrer par modélisation de l'écosystème les connaissances acquises concernant les répercussions de l'élevage de bivalves sur la structure et la fonction de l'écosystème pour évaluer l'impact net des activités* *aquacoles sur les principales composantes de l'écosystème et aborder les enjeux de la capacité de production et de la durabilité de l'aquaculture.*

- État de l'écosystème : *élaborer un système de classification et d'évaluation de l'état de fonctionnement de l'écosystème pour les régions où l'on pratique l'élevage de bivalves; intégrer aux études écosystémiques des systèmes aquacoles les multiples agents d'agression de l'écosystème qui découlent des activités humaines sur terre et en mer.*

INTRODUCTION

The bivalve aquaculture industry has expanded rapidly in Canada over the last two decades. Although highly diverse in structure, capital and material infrastructure, the most rapid development has occurred with mussel culture: an industry which, until recently, developed at an exceptional pace throughout Atlantic Canada. Ease of mussel spat collection and deployment throughout the water column, with comparatively inexpensive capital investment, have fueled the development of mussel aquaculture. In contrast, oyster, clam and scallop culture systems generally involve relatively small area operations, intertidal or bottom-culture. While significant suspended longline culture of oysters and scallops occurs in British Columbia (BC), the areas leased for these activities generally, with a few exceptions, occupy a small fraction of coastal embayments. Estuarine and coastal systems in Prince Edward Island (PEI) support 80% of bivalve culture in Canada and 98% of the total value of mussel landings in the Maritimes and Gulf Regions (www.gfc.dfo.ca), and a high proportion of suitable embayments is leased for mussel culture. The implications of such rapid and extensive water body transformation into mussel production have been recognized as having the potential for significant impact on ecological and oceanographic processes in Prince Edward Island (Shaw 1998).

Unlike finfish aquaculture, bivalve culture requires minimal additions to the environment, except for the animals themselves and the infrastructures used to grow them. Their food is supplied by the environment and their wastes return nutrients and minerals to the ecosystem. However, dense populations of bivalve filter-feeders are characterized as 'ecosystem engineers' (Jones et al. 1994; Lawton 1994), owing to their ability to modify, maintain and create entire habitats through their effects on suspended particles and the formation of structurally complex shell habitat. Suspended and bottom culture of bivalves increases the surface area available for attachment and grazing by other species, and spaces between shells provide refugia from physical stress (currents and waves) and predation (Ragnarsson and Raffaelli 1999). Potential mechanisms for ecosystem-level effects include the utilization of particulate food resources (primarily phytoplankton and detritus, but including some auto- and heterotrophic picoplankton and micro-zooplankton) by the bivalves and associated epifauna, the subsequent release of unutilized materials in dissolved (urine) and particulate (feces and pseudofeces) form, and the removal of minerals from the system in the bivalve harvest.

This paper reviews the present state of knowledge on environmental issues related to bivalve aquaculture, with particular attention given to the potential effects (both positive and negative) of suspended mussel culture. Our focus is on identifying potential changes in ecological processes (material and energy fluxes as well as nutrient cycling) at the coastal ecosystem scale (e.g. estuary or embayment) and on identifying gaps in knowledge that need to be addressed through continued research. The temporal scale addressed is primarily long-term to include ecosystem changes over seasonal, life-cycle and aquaculture site development time-scales. However, shorter time-scales are considered when important physical, chemical and biological processes have longer-term implications.

POTENTIAL ECOLOGICAL EFFECTS OF FILTER-FEEDING

Bivalve filter-feeders have a large capacity to filter water, directly altering concentrations of the seston in the surrounding water (Bayne et al. 1989; Dame 1993, 1996; Jørgenson 1996; Smaal et al. 1997). It has often been suggested that dense bivalve populations exert a strong long-term influence on energy flow at the scale of whole estuaries, bays and coastal systems by controlling phytoplankton and seston concentrations through their filter-feeding activity (Cloern 1982; Officer et al. 1982; Nichols 1985; Hily 1991; Smaal and Prins 1993; Dame 1996; Dame and Prins 1998; Prins et al. 1998). This speculation stems primarily from measurements of water clearance (filtration) rate made on individual animals that are scaled-up to predict population or community grazing capacity. Several authors have compared estimates of the time required for resident bivalve populations or communities to clear all of the water volume in their coastal system (clearance time) with the time required for the water mass to be replaced by tidal exchange (residence time) and concluded that the bivalves can exert a significant and controlling influence on particulate matter in many shallow coastal systems (reviewed by Dame 1996; Dame and Prins 1998). A similar comparison, based on estimates by Grant (2000) of mussel culture area, feeding rate and tidal flushing in PEI embayments, is presented in Figure 1. This analysis suggests that for 12 of the 15 embayments studied, the mussel biomass presently under culture is potentially capable of removing food particles much faster than tidal exchange is capable of replacing them, and therefore appears to control phytoplankton and seston at the coastal ecosystem scale through overgrazing. Meeuwig et al. (1998) used a different mass balance approach to model phytoplankton biomass in 15 PEI embayments and estimated that the mussel farms in six of these systems reduced phytoplankton biomass by 45% to 88%. These order of magnitude calculations are strongly suggestive that intensive mussel culture has the capacity to alter matter and energy cycling for long periods in some coastal systems.

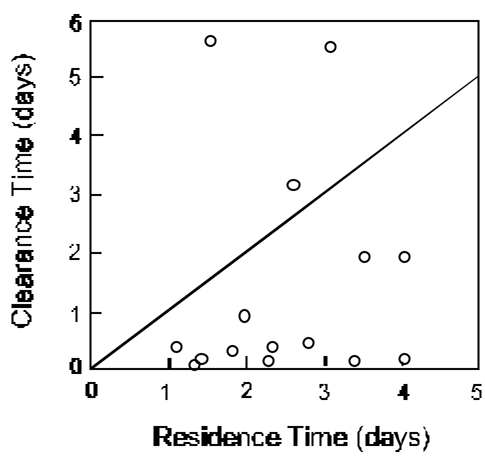

Figure 1. Comparison of predicted water mass residence time (tidal) and clearance time by mussel culture operations for 15 embayments in PEI. Mussel aquaculture potentially controls suspended particle concentrations (phytoplankton and detritus) where clearance time is less than residence time (point falls below unity line).

While simple scaling exercises, such as the one illustrated in Figure 1, are intuitive ecosystem indicators of carrying capacity and the potential impacts of existing and proposed aquaculture operations (includes biotic and abiotic factors controlling bivalve food supplies), these approaches neglect potentially important physical processes, such as water column stratification, mixing, and flow velocity, that could influence the effects of mussel culture operations on suspended particles. These approaches also use single flushing estimates for estuaries, when flushing is spatially dependent. Several biotic factors also need to be considered before placing too much emphasis on these results. First, comparison of water clearance and residence times does not consider replenishment of food particles within the estuary through internal primary production. Estimates of the time required for primary production within the system to replace the standing crop of phytoplankton (phytoplankton doubling time) are required before more definitive conclusions of the impact of bivalve filtration in these and other embayments can be reached (Dame 1996). Dame and Prins (1998) examined 11 coastal ecosystems and suggested that most of the systems produce sufficient phytoplankton internally to prevent overgrazing by resident bivalve populations. However, several of the systems studied, and particularly those under intense bivalve culture, require the import of food resources from outside the system to prevent particle depletion. Dowd (2000) examined a simple biophysical model that quantifies the relative roles of flushing, internal production and bivalve grazing on seston levels.

Another important consideration, when assessing the potential impact of bivalves on their and other filter-feeders' (e.g. zooplankton) trophic resources, is that bivalve grazing may directly stimulate system primary production such that algal cell removal may be compensated by an increase in algae growth rate. Factors that may contribute to this bivalve-mediated optimization of primary production are (1) increased light through reduced turbidity (assumes algae are light limited); (2) greater growth of algae through continuous grazing of older cells; (3) a shift to faster growing algae species; (4) increased rate of nutrient cycling; and (5) increased nutrient availability (Prins et al. 1995). Mesocosm studies examining the role of the clam *Mercenaria mercenaria* in controlling seston concentration indicated that a relatively low abundance of clams doubled primary production and altered the community structure of the plankton (Doering and Oviatt 1986; Doering et al. 1989). Grazing by mussels was also shown to result in increased picoplankton abundance (Olsson et al. 1992) and a shift to faster growing diatoms (Prins et al. 1995). While bivalve filter-feeders apparently contribute to optimizing primary production at relatively small temporal and spatial scales, the larger-scale significance of this interaction in natural ecosystems remains speculative.

An understanding of bivalve feeding rate is fundamental to accurately predict the role of bivalves in controlling seston availability and primary production. Mussels have been one of the most extensively studied marine organisms, but uncertainties and controversies regarding their physiology still exist that affect our capacity to accurately predict growth and the consequences of environmental variables on mussel bioenergetics (reviewed by Bayne 1998; Jørgenson 1996). Theories and models of bivalve functional responses to ambient food supplies vary widely in concept, resulting in considerable uncertainty on the actual ecological influence of dense bivalve populations (Cranford and Hill 1999;

Riisgård 2001). Controversy has been generated by the continued use of feeding rate measurements obtained in the laboratory using pure algal diets that are extrapolated to field conditions where cell types and concentrations and the presence of detritus may alter bivalve filtration and ingestion rates (Cranford 2001). Continued research is particularly needed on how the large seasonally variable energy/nutrient demands of mussels influence the uptake and utilization of naturally available food supplies (Cranford and Hill 1999). Further, genotype- and phenotype-dependent differences in marine bivalves also contribute to the large variance in feeding rate (reviewed by Hawkins and Bayne 1992), and this has yet to be considered in estimates of population clearance time.

The accuracy of some scaled-up estimates of bivalve population clearance time has been questioned based on the results of mesocosm studies (Doering and Oviatt 1986) and the use of new methodologies that permit bivalve feeding rates to be measured continuously under more natural environmental conditions than has been employed previously in the laboratory (Cranford and Hargrave 1994; Iglesias et al. 1998). Cranford and Hill (1999) used an *in situ* method to monitor seasonal functional responses of sea scallops (*Placopecten magellanicus*) and mussels (*Mytilus edulis*) and suggested that the coupling of coastal seston dynamics with bivalve filter-feeding activity may be less substantial than previously envisaged. That study confirmed previous results indicating that bivalves in nature do not always fully exploit their filtration capacity, but generally feed at much lower rates (Doering and Oviatt 1986). Prins et al. (1996) and Cranford and Hill (1999) showed that *in situ* and field measured clearance rates that use natural diets are similar and provide accurate predictions of bivalve growth. While it is, therefore, possible to scale up from individual measurements to bivalve populations, feeding behavior has also been shown to vary greatly over short- to long-time scales owing to external (variable food supply) and internal (variable energy demands of reproduction) forcing (Cranford and Hargrave 1994; Bayne 1998; Cranford and Hill 1999). The common practice of using average clearance rates for calculating population influences on phytoplankton may give equivocal results for much of the year.

Perhaps the best indication of the potential for bivalve filter-feeders to control suspended particulate matter at the ecosystem scale comes from observations of ecosystem changes that occurred after large biomass variations in natural bivalve populations, as well as the observed density-dependent effects of intensive cultivation practices. Population explosions of introduced bivalve species in San Francisco Bay and dramatic reductions in oyster populations in Chesapeake Bay have been implicated as the cause of the large changes in phytoplankton biomass and production experienced in these systems (Nichols 1985; Newell 1988; Nichols et al. 1990; Alpine and Cloern 1992; Ulanowicz and Tuttle 1992). Numerous similar examples can be drawn from the limnology literature with respect to the introduction, rapid growth and effect of zebra and quagga mussels (*Dreissena spp.*) on the water column in the Laurentian Great Lakes. Research on the whole-basin environmental effects of intense mussel and oyster aquaculture in the Bay of Marennes-Oléron, the most intensive growing region of the Atlantic coast of France, has focused on the impact of bivalve overstocking on growth and survival (Héral et al. 1986; Héral 1993). Intensive bivalve culture operations led to large-scale growth reduction and high mortalities in the Bay on two occasions. The large biomass of scallops under culture in Mutsu Bay, Japan also resulted in growth reduction and high mortality (Aoyama 1989). These impacts of intensive aquaculture appear to result in a feedback on bivalve growth from bivalve-induced changes in particulate food abundance and quality.

POTENTIAL ECOLOGICAL EFFECTS OF BIODEPOSITION

An important issue related to particle consumption by bivalve filter-feeders is the resulting repackaging of fine suspended material into larger feces and pseudofeces. Bivalves effectively remove natural suspended matter with particle sizes greater than 1 to 7 µm diameter, depending on species, and void them as large fecal pellets (500-3000 µm) that rapidly settle to the seabed, especially under conditions with slow or poor water flushing and exchange. This particle repackaging diverts primary production and energy flow from planktonic to benthic food webs (Cloern 1982; Noren et al. 1999). While the dynamics of bivalve feces deposition (settling velocity, disaggregation rate and resuspension) are poorly understood, enhanced sedimentation under shellfish culture is well documented (Dahlback and Gunnarsson 1981; Tenore et al. 1982; Jaramillo et al. 1992; Hatcher et al. 1994). Furthermore, mortality and fall-off of cultured bivalves, induced by seasonal colonization by fouling organisms that use suspended bivalves and their lines as substrate, can result in additional acute benthic organic loading.

Sediment organic enrichment effects are generally believed to be less dramatic with bivalve culture than with finfish culture where uneaten and partially digested food is deposited on the seabed (Kaspar et al. 1985; Baudinet et al. 1990; Hatcher et al. 1994; Grant et al. 1995). However, the zone of influence may be larger with bivalve aquaculture, if a large fraction of the total volume of coastal embayments is under culture and if hydrographic conditions permit the deposition and accumulation of biodeposits. Bivalve culture occupies a very significant portion of many embayments in PEI (mussel lease volume averaged 36% of total estuary volume for eight major PEI embayments) (Grant et al. 1995), but this is rare in other parts of Canada.

Organic enrichment of the seabed under suspended bivalve culture is due to the increased vertical flux of naturally occurring particles (Barranguet et al. 1994; Hatcher et al. 1994; Stewart et al. 1998). The seasonal biodeposition rate and organic content of fecal pellets was measured for scallops (*P. magellanicus*) and mussels (*M. edulis*) in two coastal regions in Nova Scotia (NS) (Cranford and Hill 1999), and organic matter biodeposition was observed to reach maxima in the spring and fall. That study showed that the daily biodeposition rate of a cohort of 25 mussels (80 mm shell length) increased natural sedimentation rates (g dry weight$\cdot m^{-2}\cdot day^{-1}$) by an average factor of 26 (mean of 160 daily biodeposition and sedimentation measurements). Fecal pellet organic content ranged from 20% to 70% with the highest values observed during the spring phytoplankton bloom. Feces generally had a similar organic content as other settled particles (Cranford and Hill 1999), despite containing partially digested organic matter.

If organic biodeposition by bivalves is sufficiently high, decomposition of organic biodeposits can increase the oxygen demand in sediments and generate an anaerobic environment that promotes ammonification and sulfate reduction. This is the classic response of sediments to eutrophication (Cloern 2001). An increase in benthic sulfate reduction has been observed under some intensive mussel culture sites (Dahlback and Gunnarsson 1981; Tenore et al. 1982) but not under others (Baudinet et al. 1990; Jaramillo et al. 1992; Grant et al. 1995; Chamberlain et al. 2001). Benthic responses to increased organic enrichment under suspended bivalve culture include increases in phytopigments, bacterial abundance and meiofauna community structure and biomass (Dahlback and Gunnarsson 1981; Mirto et al. 2000) and localized reductions in macrobenthic infaunal abundance and/or diversity (Tenore et al. 1982; Mattsson and Linden 1983; Kaspar et al. 1985; Stenton-Dozey et al. 1999; Chamberlain et al. 2001). These community impacts appear to be long-term, as little recovery of disturbed communities was observed 18 months after mussels were harvested (Mattsson and Linden 1983) and four years after an intensive mussel raft culture operation was removed (Stenton-Dozey et al. 1999). Although common in Europe and the northeastern United States, the raft culture technique is not utilized in Canada.

The pattern of enrichment effects can be observed in data from a survey of PEI inlets during 1997 (Shaw 1998). Redox potential (Eh), total $S^{=}$ and organic matter (OM) were measured in sediment collected at active lease sites, in adjacent reference areas away from mussel lines and in culture-free inlets where no mussel aquaculture occurred (Figure 2). The three geochemical variables have been shown to be indicators of benthic enrichment due to increased organic matter loading in areas of intensive finfish aquaculture (Hargrave et al. 1997). Significant ($p < 0.05$) differences in Eh, total $S^{=}$ and OM occurred between the three types of sampling sites. The most negative redox potentials (indicative of more anoxic conditions due to enhanced OM deposition) occurred in sediments under mussel lines. Concentrations of total $S^{=}$ and OM were not significantly different at lease and reference sites, but both of these variables were significantly higher than at culture-free sampling locations. The similarity in total $S^{=}$ and OM at lease and reference sites and differences in Eh between sampling locations implies that intensive mussel culture in PEI has had inlet-wide benthic impacts that are observable using sediment geochemical measurements.

The degree of benthic impact is expected to differ greatly between culture sites depending on the type and extent of culture activities and local environmental conditions. Observations of organic enrichment impacts from bivalve culture in PEI are not generally applicable to bivalve culture sites in other regions of Canada. The sedimentation patterns and dispersion of bivalve biodeposits are controlled by water depth and local water movement. Slight differences in these physical properties appear to explain the marked differences in the degree of impact observed on seabed geochemistry and communities under different suspended mussel culture sites (Chamberlain et al. 2001). Many embayments in PEI are also already stressed by similar eutrophication effects from land-use (see section on Aquaculture Interaction with Land-use), while culture activities in other regions tend to occur in areas with much lower agricultural nutrient inputs.

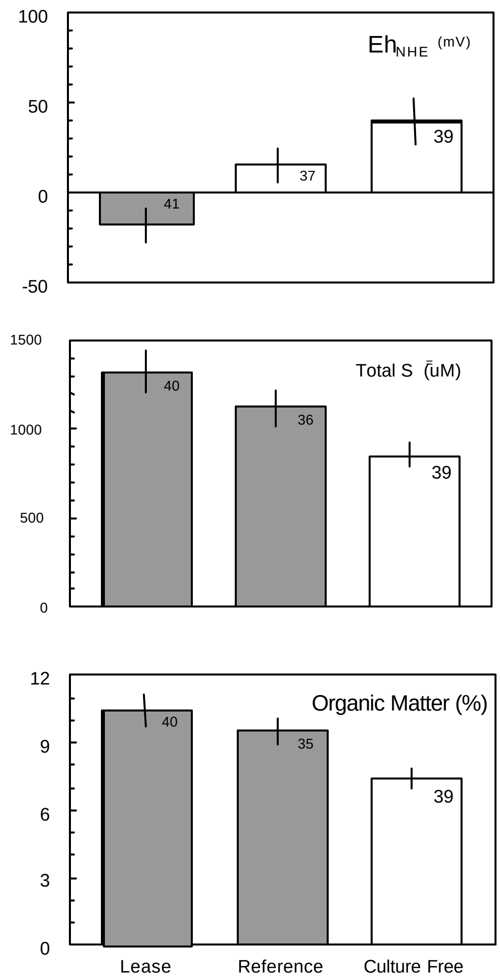

Figure 2. Data from Shaw (1998) summarizing mean (±SE) values for redox potentials (Eh), total $S^{=}$ and percent organic matter in surface sediment (2-4 cm depth layer) from 20 inlets in PEI. Samples were collected during late summer 1997 at stations under mussel lines on lease sites (Lease), at reference sites in the same inlet but >50 m away from mussel lines (Reference) and in inlets where mussel culture had not previously occurred (Culture Free). Numbers indicate pooled sample sizes. Differences in shading indicate significant differences (Mann Whitney U test, $p < 0.05$) between variables grouped by location.

Grant (2000) developed an approach for addressing the capacity of tidal action to redistribute materials deposited by mussel aquaculture operations in PEI estuaries. This modelling effort consists of estimating the balance between mussel egestion rate and the rate of tidal flushing. The aim was not to predict biodeposition effects, but to estimate the potential for whole coastal systems to resist organic loading through physical exchange processes. Estuaries identified as having the greatest risk of biodeposition effects had a relatively small tidal exchange and high percentage of the total estuarine volume under culture. There is little information available on the capacity of coastal ecosystems to assimilate organic loading and, subsequently, to resist biodegradation. Therefore, further research is needed.

The coupling of planktonic and benthic food webs, caused by the bivalves modifying, repackaging and increasing the sedimentation rate of fine suspended particles, changes the flow of energy in the ecosystem by altering the availability of food resources to other species. Crabs and demersal fish appear to benefit from culture activities as a result of the increased food availability from the fall-off of mussels and epibionts from lines (Lopez-Jamar et al. 1984; Freire et al. 1990). However, grazing competition with mussel culture can affect zooplankton and larval fish dependent on suspended seston as food. Bivalve filter-feeders have a huge competitive advantage over zooplankton, as they may significantly reduce the abundance of micro-zooplankton (<200 µm) (Horsted et al. 1988) and meso-zooplankton (up to 6 mm) (Davenport et al. 2000) through ingestion and are capable of immediately responding to increased food availability (e.g. phytoplankton bloom). The zooplankton must go through a complete life cycle before they can begin to fully exploit new resources. Mesocosm studies indicate that *Mercenaria* (infauna) and *Mytilus* (suspended culture) populations can alter pelagic food webs by suppressing the zooplankton (Horsted et al. 1988; Doering et al. 1989). Competitive pressure on zooplankton also comes from the periodic presence of large populations of cultured species larvae. The decline of oyster populations in Chesapeake Bay has been implicated in the observed increase in abundance of zooplankton and their major predators (Newell 1988). However, these potential effects on zooplankton communities are largely speculative, as they have never been documented in field studies.

CHANGES IN NUTRIENT DYNAMICS AND POTENTIAL CONSEQUENCES

The consumption and deposition of suspended particulate matter by farmed bivalves can play a significant role in controlling the amounts and forms of nitrogen in coastal systems and the rate of nitrogen cycling (reviewed by Dame 1996). This translocation of matter can provide a means of retaining nutrients, trace elements and contaminants in coastal areas where they are recycled within detrital food chains, rather than being more rapidly exported (Jordan and Valiela 1982). Benthic nutrient mineralization can increase at culture sites as a result of the increased organic matter sedimentation greatly increasing rates of nitrogen cycling (Dahlback and Gunnarsson 1981; Kaspar et al. 1985; Feuillet-Girard et al. 1988; Barranguet et al. 1994; Grant et al. 1995). Chlorophyll and nutrient mass-balance calculations for PEI estuaries show a tight correlation between phytoplankton biomass and nutrients, suggesting that nutrient availability in these intensively cultured systems primarily limits ecosystem productive capacity (Meeuwig et al. 1998). Nutrient cycling rates and availability may be increased at mussel farms through the mineralization of the large amounts of feces and pseudofeces trapped within the mussel socks. This permits nutrients to be released at shallower, more nutrient depleted depths than occurs if the nutrients are regenerated in the sediments. Decomposition of organic matter in aerobic surface sediments aids in recycling nutrients back to the water column for uptake by phytoplankton, while anaerobic decomposition in sediments under conditions of excessive organic enrichment (e.g. biodeposition) results in the production of nitrogen gas that may increase nitrogen limitation within the system. Conversely, phosphorus release from sediments is promoted under anaerobic conditions (Nixon et al. 1980).

An additional ecosystem consequence of bivalve aquaculture potentially stems from the transformation of much of the ingested particulate minerals into dissolved nutrients that are excreted as a necessary part of bivalve metabolic processes. The high flux of ammonia from dense bivalve populations appears to exert a controlling influence on nitrogen concentrations in some coastal regions (Dame et al. 1991; Strain 2002), and this aspect of bivalve culture may have a major positive effect on the phytoplankton (Maestrini et al. 1986; Dame 1996). There is little information available on the relative importance on ecosystem nutrient availability of the direct transformation of suspended particulate matter (excretion) into nutrients compared with nutrients supplied as a result of particulate matter translocation (biodeposition and remineralization) by bivalves. However, mineralization of biodeposits appears to be a more important nutrient source for phytoplankton production than direct excretion (Asmus and Asmus 1991; Prins and Smaal 1994).

While the greater availability and faster cycling of nutrients in aquaculture systems can lead to enhanced production of phytoplankton and seagrass (Peterson and Heck 2001), these changes may also contribute to more frequent algal blooms, including those of the domoic-acid-producing diatom *Pseudo-nitzschia multiseries* (Bates 1998; Bates et al. 1998). Domoic acid production is enhanced 2- to 4-fold when *P. multiseries* is grown in the presence of high concentrations of ammonium (220-440 µM) relative to the same concentration of nitrogen in the form of nitrate (Bates et al. 1993). Observed aquaculture-induced changes in the relative concentrations of silica, nitrogen and phosphorus (e.g. Hatcher et al. 1994) may also favor the growth of harmful phytoplankton classes (Smayda 1990). Impacts of changing nutrient ratios on phytoplankton community composition, including the promotion of harmful algal blooms such as *Pseudo-nitzschia*, have been documented in relation to coastal eutrophication (e.g. Parsons et al. 2002), but a causative connection has yet to be proven rigorously (Cloern 2001). Similarly, no definitive conclusions can be drawn from the sparse literature on the scale of aquaculture impacts on microalgae community composition.

The retention and remineralization of limiting nutrients in coastal systems is necessary to sustain system productivity. Benthic filter-feeders promote the retention and recycling of nutrients within coastal ecosystems by storing assimilated minerals as tissue biomass that is released upon death and decomposition (Dame 1996). Kaspar et al. (1985) suggested that the harvesting of cultured mussels may lead to nitrogen depletion and increased nutrient limitation of primary production. However, ecosystem-level effects resulting from the removal of nutrients stored in the cultured biomass are largely speculative, and further studies are needed to examine the consequences to the marine food web of nutrient removal.

AQUACULTURE INTERACTIONS WITH LAND-USE

Any attempt to assess ecosystem-level effects of bivalve aquaculture must consider the complexity of natural and human actions in estuarine and coastal systems. Ecosystem responses to multiple stressors (contaminants, fishing activities, invasive species, habitat loss, climate change, coastal construction, etc.) are intimately connected (Cloern 2001).

The determination of the cumulative effect of all human activities on coastal ecosystems is difficult but essential for environmental assessments. The capacity of cultured mussels to alter and control food supplies, energy flow and nutrient cycling depends on how other stressors positively or negatively influence important bivalve physiological processes (clearance rate, digestive efficiency, biodeposition rate and ammonia excretion) and growth. For example, infectious diseases associated with intense bivalve culture, as well as exposure of cultured organisms to 'exotic' pathogens introduced with seed or broodstock, can have a significant and frequently acute and permanent impact on the organisms' physiological and nutritional status (Banning 1982; ICES 1995; Bower and McGladdery 1996; Hine 1996; Renault 1996; Minchin 1999; Miyazaki et al. 1999). As a mostly sessile component of an ecosystem, bivalves play a sentinel role, acting as a sponge for many of the components actively or passively added to its aquatic surroundings (Dewey 2000). Important biochemical, cellular, physiological and behavioral changes in bivalves occur with contaminant exposure, and these can affect populations and disrupt energy flow and the cycling of materials within coastal ecosystems (Capuzzo 1981).

Land-use practices that result in nutrients being transported into estuaries can be a major determinant of coastal water quality and eutrophication (Chapelle et al. 2000). Concentrations of nitrogen and phosphorus in PEI estuaries have increased substantially between the 1960s and 1990s, and 10 of the 20 embayments sampled in 1998 and 1999 exhibited nitrogen levels exceeding the threshold for eutrophic conditions (DFO 2000). The large influence of agricultural activities on PEI embayments was indicated by the close correlation between chlorophyll biomass and the area of the watershed over which agriculture extends (Meeuwig 1999). Speculations that intense mussel culture can affect coastal ecosystems in positive ways by reducing eutrophication have been supported by observed changes in estuarine ecosystems in which natural bivalve populations have either dramatically increased (e.g. San Francisco Bay: Cloern 1982; Officer et al. 1982) or decreased (e.g. Chesapeake Bay: Newell 1988). Both of these systems are highly eutrophic, owing to intense farming and industrial/residential development within their watersheds.

Bivalve filter-feeders in these and other estuaries are believed to mitigate eutrophic trends by ingesting large quantities of algae and suspended particulate matter. However, this suggestion has not been proven rigorously and is based primarily on scaled-up bivalve filtration rates that may have been overestimated (Cranford and Hill 1999) and on mass balance calculations (Meeuwig et al. 1998). Asmus and Asmus (1991) suggested that the ability of mussel beds and culture sites to reduce the standing stock of phytoplankton is unlikely to combat anthropogenic eutrophication because they also promote primary production and accelerate the turnover of phytoplankton through their effects on nutrient cycling. As noted above, intense shellfish farming also increases the retention of nutrients within coastal systems (see also review by Cloern 2001), further focusing the negative effects of nutrient loading on this region. While elevated phytoplankton levels have a clear benefit to aquaculture farm productivity, the accompanying increase in organic biodeposition rates (i.e. bivalves augment pelagic/benthic coupling) could stimulate benthic microbial metabolism, alter sediment chemistry and increase the probability that benthic communities, which are highly sensitive to eutrophication, will change. Eutrophic conditions can also depress bivalve physiological functions and growth through exposure to toxic algal blooms (Chauvaud et al. 2000), limiting their perceived grazing control on algae biomass. Conversely, the removal of nutrients from the system in the bivalve harvest may help to alleviate some of the eutrophication problem. Interactions in the coastal zone between farmed bivalves and the environmental consequences of nutrient loading are highly complex, and all aspects need to be addressed objectively and integrated quantitatively before any conclusions can be reached on whether or not bivalve farming has a net positive or negative result on ecosystem quality.

Sediment released into coastal waters during land-use has the potential to alter physical habitats and directly impact marine organisms, including cultured species. There are limited quantitative data available on the effect of agriculture run-off on substrate composition and suspended sediment concentrations in PEI waterways, but anecdotal observations indicate high suspended concentrations during rainfalls and an increasing proportion of bottom covered by fine sediments (DFO 2000). Cultured bivalves and their support structures could alter sedimentation patterns within embayments by altering flow dynamics with the net result being a tendency towards accelerated deposition of fine-grained sediment. With the exception of raft culture, little is presently known about how suspended culture alters water flow (Grant and Bacher 2001), but the impact on sedimentation patterns will likely be dependent on culture spacing and local hydrographic conditions. Sediment deposition, resuspension and transport are governed in the marine environment by particle aggregation processes, which effectively control the settling velocity of fine-grained sediment by orders of magnitude. If bivalve cultures influence the natural equilibrium among the major factors controlling aggregation rate (particle concentration, particle stickiness and turbulence) (Hill 1996), sedimentary conditions within a bay may be altered.

Pesticides have been detected in 75% of stream water samples collected in PEI between 1996 and 1999 (DFO 2000). While concentrations were well below acute lethal concentrations (rainbow trout LC_{50} values), there were 12 fish kills downstream from potato fields in 1994 to 1999 that were suspected, or shown, to be caused by pesticides (DFO 2000). There is also increasing concern over the endocrine disrupting potential of released pesticides, as well as possible links between exposure of bivalves to contaminants and the incidence and severity of bivalve diseases (Coles et al. 1994; Pipe and Coles 1995; Pipe et al. 1995, 1999; Anderson et al. 1996, 1998; DaRos et al. 1998; Kim et al. 1999). The principal mechanism by which dissolved contaminants are transported in the marine environment is by scavenging (uptake) onto particulate matter and particle settling. This particle-reactive nature of organic contaminants increases their availability for filter-feeders, including wild and cultured bivalves. Bivalves bioaccumulate many abiotic contaminants and, as a result, have been widely used since the 1970s as sentinel organisms for monitoring such contaminant levels. In fact, many 'Mussel Watch' experiments have used suspended mussels in cages or other infrastructures to monitor contaminant drift in plumes, a holding mechanism akin to suspension culture (e.g. Salazar and Salazar 1997). Mussels are also used to monitor changes in environmental quality by combining and linking measurements of chemical inputs and concentrations in tissues with a pollution stress response called 'scope for growth' (SFG). SFG integrates physiological responses that affect changes in growth rate and has successfully been used to detect, quantify and identify the causes and effects of pollution (e.g. Widdows et al. 1995). Bivalve clearance rate is a component of the SFG equation and is highly sensitive to contaminant stress (Donkin et al. 1989; Widdows and Donkin 1992; Cranford et al. 1999). Although largely speculative, reduced feeding rates associated with exposure to contaminants (e.g. simultaneous nutrient and contaminant loading from agriculture) could influence their perceived capacity to mitigate coastal eutrophication by reducing their influence on ecosystem energy flow and nutrient cycling.

Although the rapid breakdown of agricultural pesticides and herbicides in water may seem to negate their significance in impacting bivalves and other aquatic organisms, there is growing concern and evidence that even the transient passage of the chemicals themselves (acute exposure) or the chronic exposure to their breakdown products may play a role in long-term or sub-acute effects. This complicates correlation to point-source or wider influent effects and makes 'mystery mortalities' difficult to resolve. This conundrum has recently gained a higher profile as a knowledge gap, especially with respect to molluscs, due to growing evidence that bivalve neoplasias appear to show strong correlations to heavily contaminated environments. Elston et al. (1992) summarized a long list of numerous neoplasia triggers that have been and are associated with bivalve neoplasias. These include pesticides, herbicides, organochlorides (Farley et al. 1991; Craig et al. 1993; Gardner 1994; Harper et al. 1994; van Beneden 1994; Dopp et al. 1996; Strandberg et al. 1998), retroviruses (Appeldoorn and Oprandy 1980; Oprandy et al. 1981; Cooper and Chang 1982; Cooper et al. 1982; Farley et al. 1986; Sunila and Farley 1989; Sunila and Dungan 1991; House et al. 1998), senescence (Bower 1989; Bower and Figueras 1989) and natural environmental extremes, such as changes in water temperature (Brousseau 1987; Brousseau and Baglivo 1991a,b; McLaughlin et al. 1996). The species that are most susceptible to neoplastic diseases are mussels (*M. edulis* and *M. galloprovincialis*) and clams (*Mya arenaria* and *Mercenaria* spp.). Blue mussels have had acute outbreaks of haemic neoplasia (blood cell dysfunction and proliferation) along the northwest coast of the United States and southern BC (Bower 1989). A correlation to water quality was not apparent. However, severe outbreaks of haemic neoplasia have been found in soft-shell clams from Chesapeake Bay, New Bedford Basin (Massachusetts) and, more recently, along the north shore of PEI (McGladdery et al. 2001). All these areas are subject to high agricultural run-off or organochloride industrial waste (Craig et al. 1993; Dopp et al. 1996; Strandberg et al. 1998). In addition, samples of the same species, collected from the Sydney tar ponds, NS, also showed levels of the condition in significant excess of 'normal' levels (McGladdery et al. 2001).

Another neoplasia condition that affects both hard- and soft-shell clams is gonadal neoplasia. The germinal cells proliferate without undergoing meiosis or differentiating into sperm or ova (Barber 1996; van Beneden et al. 1998). This condition shows a distinct geographic focus of infection, with rare outlying distribution spots. A hot spot in northern Maine shows a close correlation to forestry pest control programs, coinciding with spring warm up and gametogenesis, but there is no such correlation evident with another hot spot in southern New Brunswick (Gardner et al. 1991; Barber and Bacon 1999). Bivalve neoplasias, whether in cultured or wild populations, can be triggered by many different factors, including natural and anthropogenic causes.

Habitat degradation is well documented as having the potential to adversely affect bivalve health (Croonenberghs 2000; Dewey 2000; Moore 2000). For example, the ciliostatic properties of many *Vibrio* species (ubiquitous marine and estuarine Gram-positive bacteria) is well documented (DePaola 1981; Brown and Roland 1984; Nottage and Birbeck 1986; Nottage et al. 1989; DePaola et al. 1990). Although not demonstrated as being a factor in open-water (Tubiash 1974), the effects of these exotoxins on the ciliated larval stages of bivalves have been proven for numerous species under hatchery-rearing conditions (Tubiash et al. 1965, 1970; Elston et al. 1981, 1982, 2000; Elston 1989; Nicolas et al. 1992). Severity of infection is most commonly related to sub-optimal growing conditions (accumulation of dead or dying larvae, contaminated algal food, residual gametes, etc.) that enhance bacterial proliferation and compromise the immune responses of infected larvae (Elston 1989). Sensitivity to *Vibrio* spp. can vary considerably. Sindermann (1988) cites 10^2 *vibrio* cells·ml^{-1} as being potentially pathogenic to oyster larvae, while other bivalves can tolerate 10^5 cells·ml^{-1} (Perkins 1993). There is, therefore, a strong likelihood that chronic or acute blooms of these bacteria under open-water conditions could have a deleterious effect on bivalve larval recruitment, especially under conditions of warm water, rainfall and bivalve spawning (DeLuca-Abbott et al. 2000; Herwig et al. 2000). In addition, the effects of ciliostatic toxins on the ciliated digestive tracts of adult bivalves cannot be overlooked. At least two shell-deforming conditions in juvenile oysters and juvenile to adult clams have been linked to bacteria. 'Brown ring disease' of *Tapes* spp. in Europe is caused by a new *Vibrio* species, *V. tapetis* (Borrego et al. 1996; Castro et al. 1997; Novoa et al. 1998; Allam et al. 2000), and juvenile oyster disease of American oysters (*Crassostrea virginica*) is caused by a novel alpha-proteobacterium (Boettcher et al. 1999, 2000). Both these bacteria appear to proliferate in estuarine conditions and elicit energetically-costly defense mechanisms in the bivalves that are manifest in conchiolin deposition around the mantle margins. Histological profiles of the epithelial tissues of the mantle and digestive system have also shown extensive haemocyte infiltration, indicative of physiological stress (Plana and LePennec 1991; Allam et al. 1996). The linkage of these bacteria to overall habitat quality has yet to be determined.

Another set of recent studies has focused on immunosuppression induced in bivalves exposed to heavy metals and hydrocarbon-based chemical waste. The effect of these chemicals is complex, and initial results show a potential for hormesis (lower concentrations suppress haemocyte-mediated defence activities and greater concentrations show a neutral or increase in phagocytic activity), both ends of which have energetic costs to the bivalve (St-Jean 2002a,b). If these results are extrapolated for chronic, sub-lethal effects, some studies using scope for growth as a measure for carrying capacity may need to be revisited. This applies equally to the neoplasia conditions discussed above. Mortality and weakening due to infectious disease is relatively easy to quantify and correlate to environmental factors (epidemiology of the disease). However, immunosuppression and carcinogenic effects are more insidious and could readily be masked by or distort other more obvious environmental correlations. This is also important for assessment and interpretation of bivalve aquaculture impacts on environmental conditions. Weakening, impeded feeding and filtration activity, along with spawning failure or poor quality spawn can all contribute to morbidity, mortality and fall-off, with the environmental consequences discussed above.

There have been attempts to bring the effect of infection status on overall physiological performance of bivalves into bilateral correlations between physiological scope for growth and environmental carrying capacity as well as contamination, but such studies are rare and inconclusive (DaRos et al. 1998). Conceptual models of interactions between bivalve culture activities, eutrophication and ecosystem functioning are more rapidly evolving (Cloern 2001). But gaps in knowledge need to be addressed on how these and other stress components work together, if we are to broaden our understanding of cumulative environmental effects in the context of aquaculture.

INTEGRATION OF AQUACULTURE/ENVIRONMENT INTERACTIONS

A mechanistic understanding of coastal ecosystem functions is fundamental for formulating management strategies. The study of aquaculture ecosystems requires consideration of biological, physical, chemical and geological factors. Important biological processes include mussel feeding and egestion, as well as the dynamics of the supporting planktonic ecosystem and interactions with the benthic community. Physical processes governing water motion and mixing determine the transport and supply of dissolved and particulate matter. Nutrient dynamics and cycling depend on the transformations mediated by the various ecosystem components including bacteria. The sedimentation of particles is governed by the competing processes of flocculation and turbulence. These areas require research involving field measurements as well as comprehensive modelling studies that integrate available knowledge about natural- and human-driven parts of coastal ecosystems.

Coastal waters where aquaculture is practiced exhibit a variety of physical oceanographic processes (e.g. tidal and estuarine circulation). Dissolved and suspended matter in the water column are transported and mixed by water motion and eventually exchanged with the adjacent open ocean or deposited (utilized) locally. A basic understanding of particle dynamics requires tracking of the total particulate load (turbidity), food particles for shellfish (chlorophyll) and the water flux (mixing and exchange) (Grant and Bacher 2001). The effects of mussels on water column and sediment properties are influenced by circulation and mixing processes. It is hypothesized that the severity of these ecosystem effects in different coastal areas is regulated by water motion and mixing. Inclusion of oceanographic parameters is essential to a quantitative assessment of the validity of this hypothesis. Aquaculture effects are believed to be greatest in estuaries and inlets where water residence time is long and mussel biomass is high. In such areas, mussel feeding could dramatically reduce the concentration and alter the nature of suspended particulate matter, with the resultant potential to change pre-culture productivity within a defined area. In areas with greater flushing, water depleted of particles by mussels can be renewed by tidal exchange and culture-generated biodeposits may be flushed from the system.

A variety of models have been applied to assess the environmental interactions of bivalve aquaculture operations (Grant et al. 1993; Dowd 1997; Grant and Bacher 1998; Smaal et al. 1998; Meeuwig 1999). While all of the approaches include a comparison of physical water exchange to some sort of biological process like filtration, there are no standard methods for assessment of ecosystem effects. Bearing in mind the complexity of interacting factors, this is not surprising. Empirical studies, such as the calculation of budgets (e.g. carbon, nitrogen and energy) and simulation modelling, have been some of the more focused approaches to evaluating potential mussel aquaculture effects at an ecosystem level. As an example of the former, Carver and Mallet (1990) calculated the mussel carrying capacity of an inlet in eastern Canada by comparing estimated food demand to food supply based on organic seston concentrations delivered by a simple tidal prism model. The latter approach was used by Raillard and Menesguen (1994), who constructed a simulation model for a macrotidal estuary in France to describe relationships between oyster feeding, primary production and seston transport. Both approaches yield different, but complimentary, information.

Numerical models are powerful tools to help guide coastal ecosystem management because they integrate the important processes that represent this system complexity (Cloern 2001). The use of models also provides an excellent means to identify gaps in knowledge. Simulation models may be the most practical way to assess the potential net negative effect of mussel grazing on phytoplankton and zooplankton abundance and the potentially positive effect of increased remineralization on primary production (Fréchette and Bacher 1998). Similarly, ecosystem modelling can be used to quantitatively assess the contribution of cultured bivalves in combating eutrophication and of the ecological importance of nutrient losses in the mussel harvest. Fully coupled biological-physical models may be envisioned (e.g. Prandle et al. 1996; Dowd 1997) that predict ecosystem changes in chlorophyll, nutrients and other variables of interest as a function of culture density and location. To do this, shellfish ecosystem models, including carrying capacity models, must be integrated with information on water circulation, mixing and exchange to account for transport and spatial redistribution of particulate and dissolved matter. Box models (Raillard and Menesguen 1994; Dowd 1997; Chapelle et al. 2000) offer a practical means to couple coastal ecosystem models with physical oceanographic processes. The bulk parameterizations of mixing required for these box models can be derived directly from complex hydrodynamic models (Dowd et al. 2002). One interesting feature of the ecosystem model of Chapelle et al. (2000) is that the ecosystem effects of shellfish are incorporated by prescribing their biomass levels and, thereby, their effect of grazing and nutrient generation on the ecosystem, while avoiding the inclusion of mussel bioenergetic relations in detail. A promising avenue for improving ecosystem models is the use of inverse, or data assimilation, methods (Vallino 2000). These systematically integrate available observations and models, thereby combining empirical and simulation approaches, and improve predictive skill.

Simulation models that focus on estimating mussel carrying capacity and related ecosystem impacts provide effective tools for quantitative descriptions of how food is captured and utilized by mussels, as well as site-specific information on ecosystem variables and processes (Carver and Mallet 1990; Brylinsky and Sephton 1991; Grant 1996). An increased understanding of mussel feeding rates and efficiencies (ecophysiology) is fundamental to most model-based predictions of ecosystem effects, as the bivalve functional response is the basis for potential interactions between bivalves and the ecosystem. The ability to predict physiological responses of bivalves under culture conditions permits calculation of clearance, biodeposition and growth rates, and this ability presents tremendous opportunities to manage the sustainability of the industry (Carver and Mallet 1990; Labarta et al. 1998). From a mathematical perspective, the nonlinear functional relationships used to describe mussel bioenergetics have often led to poor model predictions due to their high sensitivity to inadequately known physiological parameters (Dowd 1997). Robust mathematical relations are being developed with the needs of simulation models in mind, such that bioenergetic models have been successful in predicting growth (Dowd 1997; Grant and Bacher 1998; Scholten and Smaal 1998).

Validation of models with field observations ('ground-truthing') is essential. *In situ* observations indicate where models are deficient and suggest how model structure should be altered. Model simulations can, in turn, provide a focus for field efforts. A variety of oceanographic instruments exists for monitoring biological and physical processes, and include tide gauges, current meters, fluorometers and transmissometers. Their deployment in mooring mode or as towed vehicles, in the case of particle sensors, is essential for monitoring the changing environmental conditions that occur at culture sites and the influence of mussels on these conditions. They also provide important ground-truthing information for other monitoring technologies such as remote sensing (Herut et al. 1999). Additionally, collection of data with this instrumentation is vital in the construction of models to predict the transport of water and particles at culture sites (Ouboter et al. 1998).

Decision-support systems have been developed that integrate available knowledge about natural- and human-driven parts of coastal ecosystems into computer-based models (Crooks and Turner 1999; deJonge 2000). While the prediction of future ecosystem changes is largely unfeasible as ecosystems do not exist in a stable state, computer models can be used to explore the main direction of effects on ecosystem functioning that result from various culture practices (deJonge 2000), and are useful for developing general ecological principles. It should also be emphasized that the study of culture impact using simulation models and field measurements can also be directed toward assessment of mussel growth and carrying capacity from the standpoint of farm management. For instance, bivalve studies based on physical transport of food particles to mussels and their bioenergetic use by the animals are part of a growth equation including biodeposition. The intake of food used to predict biodeposition is also part of a growth equation. Coupled with estimates of stocking density, these models produce farm yields, which may then be exported to economic models of profitability (e.g. Samonte-Tan and Davis 1998). An essential feature of the growth models is that they may be fully ground-truthed using mussel harvest/growth data from the farm sites. Ultimately, these models may be used to actively manage the location and extent of culture in coastal estuaries for multiple users. Such models will need to take into account culture dynamics, such as seed-stocking and fouling biomass, depth of activity and cumulative effects of neighboring human activities (e.g. agriculture run-off, construction sedimentation, boating and ballast activities, etc.).

Another new development that must be taken into consideration for ecological modelling is increasing interest in bivalve polyculture. Mussel culture, although predominant in Atlantic Canada, is rarely conducted in isolation from other bivalve culture. Some leases accommodate mussels, oysters, clams and, more recently, scallops. All have differing physiologies and production dynamics. Accurate modelling of single-species culture interactions with surrounding habitat ecology needs to take this into account in the future. Likewise, spat collection is frequently a 'hit and miss' operation, trying to maximize collection of the species of interest in amongst all the other bivalve species forming a continuum of production through the spring and summer spawning seasons. This further highlights the need to take the multi-species and interactive nature of bivalves into account, both within culture and pre-collection from the wild. As indicated at the start of this review, mollusc culture is much more intricately and inextricably linked to its environment than most finfish culture (even mariculture cages). Monospecific models of aquaculture interactions with habitat ecology cannot, therefore, be readily extrapolated to other bivalve species.

SYNOPSIS AND RESEARCH NEEDS

The culture of bivalve molluscs may involve a number of effects on the current state of coastal marine ecosystems. Extensive bivalve culture (suspended and benthic) has the potential for causing cascading effects through estuarine and coastal foodwebs, altering habitat structure, species composition at various trophic levels, energy flow and nutrient cycling. There have been few direct studies on the influence of mussels at the ecosystem level, but several studies have speculated on the potential for mussel cultivation to approach and even exceed the capacity of the ecosystem to maintain environmental quality (Deslous-Paoli et al. 1987; Rodhouse and Roden 1987; Asmus et al. 1990; Prins and Smaal 1990; Dame 1993, 1996). The rapid and extensive transformations of water bodies into mussel production could change the ecological function of some bays. Potential ecosystem-level effects (positive and negative) related to intensive bivalve aquaculture include the following:

- bivalve filter-feeder populations crop the resident phytoplankton so that they depend on the tidal input of offshore phytoplankton to sustain high density culture;
- large bivalve farming operations may help to reduce excess phytoplankton caused by eutrophication through grazing;
- the substitution of bivalves for zooplankton in estuaries and bays alters food webs;
- the increased sedimentation of organic matter through biodeposition acts to retain nutrients in the system;

- recycling of organic biodeposits increases the oxygen demand in sediments, generating an anaerobic environment that promotes ammonification and sulfate reduction;
- the rate of nitrogen cycling is increased through rapid deposition of organic matter, nutrient regeneration in sediments and the excretion of ammonia by mussels;
- a shortened cycle of nutrients between the benthos and phytoplankton may increase local nutrient availability as less material is exported; and
- the greater availability of nutrients leads to enhanced primary production, potentially contributing to more frequent algal blooms, including toxic species.

Few studies have been completed which adequately assess these potential environmental interactions of this newly developed industry, and few quantitative measures exist to measure ecosystem-level effects. A commonly employed means of addressing uncertainty resulting from gaps in knowledge is to establish rigorous environmental effects monitoring (EEM) programs that can provide early warning of adverse environmental effects and aid in identifying unforeseen effects (additional areas of concern). However, research is also needed to develop ecosystem-based EEM approaches and indicators that specifically address the close linkage that exists between cultured bivalves and numerous biotic (ecosystem structure and function) and abiotic ecosystem components. Development of effective EEM approaches would help to minimize the potential for exceeding system carrying capacity, while benefiting industry by optimizing farm yield.

The following research topics and associated research and development studies were identified by the authors for further study. While short-term laboratory and field studies at culture operations will be useful to address the identified gaps in knowledge, longer-term studies at new lease development sites (baseline to full development sampling) would be particularly insightful. While an immediate need for such research exists for heavily leased PEI embayments, the extensive development of the mussel industry in PEI largely precludes such studies, owing to the lack of many baseline data and difficulties in selecting the control sites needed for effective experimental designs. Such studies may be best conducted in regions where the industry is less well-developed. Intentional ecosystem manipulation experiments could also provide insights but would be both challenging and costly. Readers should note that the following separation of research topics is strictly an exercise to identify specific gaps in knowledge. The development of a mechanistic understanding of the temporal and spatial scales of ecosystem-level impacts from bivalve aquaculture requires a closely integrated multidisciplinary approach that includes major elements from each of the following research topics. Such an approach will permit even short time/small space observations to be fully utilized to address the long-term/large space issue that is the topic of this review.

1. *Ecological role of bivalve filter-feeders.* Studies are required to improve our understanding of the density-dependant role of bivalves in controlling phytoplankton and seston (including microbes) concentrations, and to determine if bivalves have a net negative (reduce standing stock) or positive (stimulate production) effect on suspended matter concentrations.

- Conduct seasonal studies of suspended particulate matter, phytoplankton biomass and primary production in estuarine and coastal systems under culture, and use the results to assess the potential for overgrazing of food resources by cultured bivalves.
- Determine the effect on suspended particulate matter, phytoplankton abundance, community structure and production of different levels of bivalve grazing pressure.
- Assess the capacity of available *in situ* and remote sensing technologies to visualize near- and far-field effects of mussel aquaculture on suspended particle fields (e.g. chlorophyll).

2. *Bivalve bioenergetics.* Given that bivalve physiological processes (feeding, respiration, biodeposition and excretion) are the primary mechanisms for potential interactions between bivalve aquaculture and the ecosystem, and therefore the sustainability of coastal operations, a more complete understanding of the physiological ecology of each species is needed to facilitate accurate prediction of ecosystem responses.
   - Identify interspecific differences in feeding and absorptive selectivity, particularly under field conditions, to quantify contributions from different food resources (e.g. retention of bacteria, differential ingestive and absorptive selection for algal species, and absorption efficiency of detritus sources) for use in carrying capacity predictions.
   - Develop robust predictive relations for the functional responses of culture species to environmentally relevant conditions.
   - Establish a clear genetic base to bivalve physiological performance.
   - Quantify the effect of the variable energy demands of gonad growth on bivalve feeding behavior.
   - Use mathematical relations for bivalve responses to internal and external forcing for continued improvement of bioenergetic models. Test growth predictions using site-specific harvest/growth data from aquaculture farms.

3. *Organic loading.* Studies are needed to determine the capacity of different coastal ecosystems to assimilate organic matter for use in predicting environmental impacts and ecosystem management.
   - Quantify organic biodeposition rates, benthic organic enrichment effects (e.g. anoxic conditions, sulfate reduction and reduced biodiversity) and recovery times at aquaculture and reference sites.
   - Study the settling and transformation of fecal wastes as a function of different physical environmental conditions.
   - Quantify the capacity of different environmental conditions to mediate organic enrichment impacts from aquaculture.
   - Develop and test surrogate measures of the total assimilative capacity of coastal systems.

4. *Nutrient dynamics*. Conduct detailed studies of nutrient dynamics in coastal systems, including those supporting and associated with bivalve aquaculture, to address the potential effects on nutrient availability and cycling.
   - Confirm nutrient limitation of phytoplankton production in coastal embayments, and identify biotic and abiotic processes contributing to nutrient limitation.
   - Document the import and export of nutrients in coastal aquaculture ecosystems, and determine the role of cultured bivalves in retaining and promoting the rapid recycling of nutrients within the system.
   - Assess the relative importance of bivalve excretion and particle biodeposition in the recycling of nutrients and the production of phytoplankton.
   - Conduct field studies to provide insights into potential interactions between nutrient dynamics and the onset of harmful algal blooms, especially those of the domoic-acid-producing diatom *Pseudo-nitzschia multiseries*.
   - Assess the potential consequences to ecosystem productivity of large nutrient losses to the bivalve harvest.

5. *Ecosystem structure*. Investigate the ecosystem-level effects of bivalve culture on ecosystem structure (abundance and biodiversity of pelagic and benthic communities) through direct competition for food resources by bivalves, zooplankton and epibionts, and the transfer of energy and nutrients to the benthic foodweb.
   - Assess the implications of reduced zooplankton abundance and composition on higher trophic levels including fish.
   - Determine the ecological role of fouling organisms (epibionts) associated with bivalve culture.
   - Investigate the ecological risk imposed by the introduction and transfer of exotic fouling and infectious agents with live shellfish transfers.
   - Investigate the ecological risk related to the potential increased incidence of infectious diseases associated with intensive culture operations.

6. *Cumulative effects.* Assess cumulative effects of anthropogenic land- and marine-use on coastal ecosystems.
   - Conduct research on the inputs and impacts of sediment, toxic chemicals, animal waste (including bacteria) and nutrients reaching embayments supporting bivalve aquaculture.
   - Assess the capacity of bivalve aquaculture to mitigate coastal eutrophication trends through their grazing on phytoplankton.
   - Investigate the effect of aquaculture on marine particle aggregation processes (particle dynamics) and the consequences to coastal sedimentation trends.
   - Conduct studies on the potential for culture activities to alter the transport and fate of particle-reactive contaminants originating from land-use.

7. *Ecosystem modelling.* Integrate knowledge obtained on the consequences of bivalve culture to ecosystem structure and function through the use and predictive power of ecosystem modelling.

- Test the ability of models to provide decision-support for the development of effective area-wide management strategies for promoting the environmental sustainability of the aquaculture industry.
- Conduct sensitivity analyses of modelled variables to assess the suitability of different ecosystem indicators for use in characterizing and monitoring ecosystem health and productive capacity.
- Develop and utilize new instrumentation and data collection strategies to obtain ecosystem data, including measurements of contaminants, for testing (ground-truthing) model predictions.
- Use models to test the hypothesis that the severity of aquaculture impacts in different estuaries is regulated primarily by water motion and mixing.

8. *Ecosystem status*. Develop indicators (methodologies and technologies) for use in aquaculture monitoring programs that provide information on ecosystem function. Test the effectiveness of selected indicators for detecting potential ecosystem-level effects of bivalve aquaculture. Identify indicator reference points that characterize ecosystem status.
    - Identify sensitive and cost-effective ecosystem health indices.
    - Establish baseline environmental conditions and the degree of natural variation in ecosystem health indices.
    - Develop a scheme for classifying the state of ecosystem functioning, including the identification of relative threshold levels.
        - Establish cause-effect relationships between culture practices (e.g. stocking density and husbandry practices) and identify candidate indicators.
        - Develop standard protocols for rapidly assessing mussel performance (growth rate, meat yield and yield per sock) at lease sites as an indicator of ecosystem impacts (i.e. impact on growth depends on impact of mussels on environment), and establish cause-effect relationships between environmental conditions and mussel performance.
        - Develop tools that incorporate information provided from ecosystem indicators that provide an integrated assessment of ecosystem status.

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CHEMICAL USE IN MARINE FINFISH AQUACULTURE IN CANADA: A REVIEW OF CURRENT PRACTICES AND POSSIBLE ENVIRONMENTAL EFFECTS

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EXECUTIVE SUMMARY

There has been a great deal of scientific debate regarding the environmental consequences of chemical usage in aquaculture. The debate has also moved into the public domain: where views of the opposing sides are typified by several highly publicized anti-aquaculture articles and, most recently, television documentaries (Ellis 1996; Goldburg and Triplett 1997; Milewski et al. 1997), and the responses to these articles from the finfish aquaculture industry (e.g. Canadian Aquaculture Industry Alliance 2001a,b).

Scientific reviews of the subject have been prepared by Zitko (1994) and GESAMP (1997). Issues raised and recommendations made by these authors have yet to be addressed in a significant manner. In addition, the authors of recent reviews of environmental impacts of aquaculture have identified chemical inputs from aquaculture activity as an area requiring further research (Nash 2001; Anonymous 2002). Several projects recently funded by Fisheries and Oceans Canada's (DFO) Environmental Science Strategic Research Fund (ESSRF) have allowed scientists to begin to address some of these topics. However, these projects are still in the early stages of identifying sources of contamination and potential effects on the environment, particularly to non-target species.

This review is a summary of potential sources of chemical contamination, chemicals that may be involved and knowledge about the potential effects of these compounds. Each identified class of chemical contaminants could be the subject of its own comprehensive review. Pesticides, drugs, persistent organic pollutants and metals are discussed in the context of the Canadian aquaculture industry.

Two classes of compounds will not require further research. Food additives include antioxidants (preservatives) and carotenoid pigments (flesh coloring) and are unlikely to cause any effects in the environment. MS-222 (tricaine methanesulfonate) is used in the New Brunswick aquaculture industry, and no adverse environmental effects are foreseen with its use (Zitko 1994).

Chemicals used in the Canadian aquaculture industry are identified in Table 1. The table summarizes recent scientific information regarding their use, persistence and potential effects in the environment. There are relatively few publications in the primary literature regarding the environmental fate and effects of chemicals used in aquaculture in Canada.

It is clear that a number of gaps in knowledge exist for each compound or class of compound. A more thorough review of each compound would identify further specific gaps related to that chemical.

For antibiotics, there appears to be no published data collected around Canadian aquaculture sites regarding the following: presence of antibiotics in sediments and aquatic biota; presence and prevalence of antibiotic-resistant organisms in sediments and indigenous species; or antibiotic residues in fish and non-target aquatic organisms. Accumulation of antibiotics in sediments may interfere with bacterial communities and affect mineralization of organic wastes (Stewart 1994), but no studies have been published in Canada.

Most work on pesticides to date has been conducted in the laboratory and has focused on determining the acute responses of aquatic organisms (non-target species) to exposure(s) to anti-sea lice chemicals. Limited field trials have focused on lethality of single treatments. Short-term responses to pesticide applications and long-term studies to establish the natural variability in local populations and measures of change in biodiversity need evaluation. Currently, commercially important non-target species have attracted much of the attention regarding effects of chemicals. There are apparently no data regarding the effects of these chemicals on microorganisms and planktonic species that form the foundation of the marine food chain in the near-shore environment. The chemical formulations of pesticide and disinfectant products have not been determined, and many of the 'inert' ingredients may be toxic to aquatic biota (Zitko 1994).

Little is known about the relationship between aquaculture and environmental contaminants, such as persistent organic pollutants (POPs) and metals. Feeds may be a source of contaminants to farmed fish. Knowledge of the constituents of each formulation is required for an accurate assessment of potential risk. Metals may be deposited near aquaculture sites from at least two other sources: leaching from metal cage structures and antifoulant paints. Chlorinated compounds (Hellou et al. 2000) and metal concentrations (Chou et al. 2002) were found to be higher when the total organic carbon content was high in sediments. Wooden cages with styrofoam floats may be a source of plastic contaminants (Zitko 1994). However, little known is known about the effects of plastics on aquatic organisms.

In addition, generic gaps can be identified in relation to the scientific approach and methodology:

- *Chemical-related research is needed in all areas where marine finfish aquaculture is practiced in Canada. Research needs to be continued in New Brunswick, where scientists have a considerable database upon which to build and have the best opportunity to monitor long-term trends. In addition, work needs to be expanded in Newfoundland, Nova Scotia and British Columbia, where little such work has been conducted.*

- *Toxicity data are limited to lethality tests conducted over short time frames (e.g. 24, 48 and 96 h). More work is required to determine chronic lethal and sublethal effects and the effects of realistic exposures of these compounds on indigenous species.*

- *While there are laboratory-derived data on many compounds, there is almost no information regarding effects of chemicals of aquaculture origin in the field. Field surveys and experiments that investigate short-term responses to chemical application as well as long-term studies to establish natural variability in local populations and measure changes in biodiversity (and other indicators of environmental health) are needed.*

- *Toxicity testing relies on single species and single compound testing in the laboratory. There is a serious lack of data regarding the cumulative effect of exposure to chemicals and the concentration and fate of chemicals of aquaculture origin. The cumulative impact of chemicals and impact of multiple exposures to non-target organisms need to be determined.*

Table 1. A Summary of Chemical Compounds Used in the Canadian Aquaculture Industry**

Chemical	Use	Persistence in Sediment	Bioaccumulation	Potential Effects
Oxytetracycline	Antibiotic	Persistent for long periods depending on environmental factors (Björklund et al. 1990; Samuelsen 1994; Hektoen et al. 1995; Capone et al. 1996); Half-life 419 days under stagnant, anoxic conditions (Björklund et al. 1990)	Uptake by oysters and crabs either in the laboratory or in close proximity to salmon cage sites (DFO 1997); Concentration in tissues of rock crabs over US FDA limit (Capone et al. 1996)	Resistance to oxytetracycline may occur in fish, non-target organisms and bacterial community near aquaculture sites (Björklund et al 1991; Hansen et al. 1993; Hirvelä-Koski et al. 1994)
Tribrissen	Antibiotic	Estimated half-life of 90 days at 6-7 cm deep (Hektoen et al. 1995)		
Romet 30	Antibiotic		Uptake by oysters (Jones 1990; LeBris et al. 1995; Capone et al. 1996; Cross unpublished data)	
Florfenicol	Antibiotic	Estimated half-life of 4.5 days (Hektoen et al. 1995)		
Teflubenzuron	Drug; In-feed sea lice control	Solubility 19 $\mu g \cdot L^{-1}$ with a log K_{ow}[a] of 4.3, indicating a potential to persist (Tomlin 1997); Persistence >6 months in area <100 m from treated cage (SEPA 1999b)		Chitin formation inhibitor; Juvenile lobster mortalities reported (SEPA 1999b); Mitigation possible by depuration prior to molting (McHenery 1997; SEPA 1999b)
Emamectin benzoate	Drug; In-feed sea lice control	Solubility 5.5 $mg \cdot L^{-1}$ with log K_{ow} of 5, indicating potential to persist (SEPA 1999b)	Withdrawal period of 25 days prior to marketing salmon	Chloride ion movement disruptor (Roy et al. 2000); Lethal to lobsters at 735 $\mu g \cdot kg^{-1}$ of food (Burridge et al. 2002); Induces molting in lobsters (Waddy et al. 2000c)
Ivermectin	Drug; In-feed 'off-label' treatment for sea lice control	Solubility of 4 $mg \cdot L^{-1}$ (Tomlin 1997); Could persist for 28 days (Wislocki et al. 1989; Roth et al. 1993)	Withdrawal period of 180 days prior to marketing; Accumulated in lobster tissue over 10 days (Burridge, Haya and Zitko unpublished data)	Chloride ion movement disruptor (Roy et al. 2000); Cumulative 80% Atlantic salmon mortality to 0.2 $mg \cdot kg^{-1}$ for 27 days (Johnson et al. 1993); 96-h LC50 at 8.5 $mg \cdot kg^{-1}$ food for shrimp; NOEC[b] was 2.6 $mg \cdot kg^{-1}$ food (Burridge and Haya 1993)
Azamethiphos	Pesticide; Bath treatment for sea lice control	Solubility 1.1 $\mu g \cdot L^{-1}$ with a log K_{ow} of 1.05, not expected to persist (Tomlin 1997)	Unlikely to accumulate in tissues (Roth et al. 1993, 1996)	Neurotoxin, acetylcholinesterase (AChE) inhibitor, but not cumulative (Roth et al. 1993, 1996); Mutagenic *in vitro* (Committee for Veterinary Medicinal Products 1999; Zitko 2001); 1-h bath at 1 $mg \cdot L^{-1}$: lethal to 15% salmon after 24 h (Sievers et al. 1995); Larval/adult lobster 48-h LC50 at 3.57-1.39 ìg·L^{-1} /NOEC 120 min at 1 ìg·L^{-1} (Burridge et al. 1999a, 2000a); Behavioral responses at >10 ìg·L^{-1} (Burridge et al. 2000a,b)

Table 1 (continued). A Summary of Chemical Compounds Used in the Canadian Aquaculture Industry**

Chemical	Use	Persistence in Sediment	Bioaccumulation	Potential Effects
Copper-based antifouling paints	Antifoulant; Reduce fouling biota on nets	Elevated copper (Cu) reported in sediments (Burridge et al. 1999a)	May accumulate in aquatic biota	100-150 mg(Cu)·kg $^{-1}$ in sediment may affect benthic fauna diversity (Debourg et al. 1993); Most sample locations > ISQG[c] of 18.7 mg·kg $^{-1}$, lethal to amphipods and echinoids (Burridge et al. 1999a)
Iodophors	Disinfecting equipment	Not expected (Zitko 1994)		Formulations may contain compounds harmful or toxic to aquatic biota (Zitko 1994; Madsen et al. 1997; Ashfield et al. 1998)
Chlorine/Hypo-chlorite	Disinfectant; Net cleaning			Toxic to aquatic organisms (Zitko 1994)
PCBs, PAHs,p,p"-DDE	Found in fish feed (Zitko 1994)	PCBs not detectable at 0.05-0.10 µg·g $^{-1}$ dry wt (Burridge et al. 1999a); p,p'-DDE detected at DL=1 ng·g $^{-1}$, dry wt (Hellou et al. 2000)	Changing lipid profiles in wild fish (Zitko 1994)	
Cadmium, Lead, Copper, Zinc, Mercury	From cage structures; Fish feed	Copper >2, zinc 1-2 times higher in sediments below cages than in fish feed (Chou et al. 2002); Cadmium exceeded 0.7 µg·g $^{-1}$ (Burridge et al. 1999a)	May be toxic or accumulate in aquatic biota	
Polystyrene beads	Styrofoam floats	Source of low molecular weight contaminants (Zitko 1994)		Benthic fauna altered by altering pore water gas exchange, by ingestion or by providing habitat for opportunistic organisms (Goldberg 1997)

** The table includes only compounds known to be used (presently or historically) in Canada. Other classes of compounds are used routinely in other jurisdictions and may be introduced to Canada in the future.

a – log K_{ow} = logarithm of the octanol-water partition coefficient. It is internationally accepted that log K_{ow} >= to 3 indicates a potential to bioaccumulate. The *Canadian Environmental Protection Act* recognizes log K_{ow} >= 5 as indicative of potential to persist and/or bioaccumulate (Beek et al. 2000).

b – NOEC = No Observed Effect Concentration

c – ISQG = Interim Sediment Quality Guidelines

UTILISATION DE PRODUITS CHIMIQUES EN PISCICULTURE MARINE AU CANADA : ÉTUDE DES PRATIQUES ACTUELLES ET EFFETS POSSIBLES SUR L'ENVIRONNEMENT

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RÉSUMÉ

Les répercussions environnementales de l'utilisation de produits chimiques dans la pisciculture ont fait l'objet de nombreux débats scientifiques. Ces débats sont maintenant dans le domaine public : les points de vue des parties adverses sont exprimés dans plusieurs articles anti-aquaculture très publicisés et, plus récemment, dans des documentaires télévisés (Ellis, 1996; Goldburg et Triplett, 1997; Milewski et al., 1997), de même que dans les réponses de l'industrie piscicole à ces publications (p. ex. Alliance de l'industrie canadienne de l'aquaculture, 2001a et b).

Zitko (1994) et GESAMP (1997) ont effectué une revue de la littérature scientifique sur le sujet. Les questions qu'ils ont soulevées et leurs recommandations n'ont toujours pas été dûment prises en considération. De plus, les auteurs de récents examens des répercussions environnementales de l'aquaculture ont déterminé que l'utilisation de produits chimiques dans cette industrie devait faire l'objet de davantage de recherches (Nash, 2001; Anonyme, 2002). Le Fonds de recherche stratégique en sciences environnementales (FRSSE) de Pêches et Océans Canada a récemment financé plusieurs projets qui ont permis aux scientifiques de se pencher sur certaines de ces questions. Ces derniers ne font cependant que commencer à déterminer les sources de contamination et les effets possibles sur l'environnement, particulièrement sur les espèces non visées.

Cet examen constitue un résumé des sources possibles de contamination chimique, des produits chimiques qui pourraient être en cause et des connaissances sur les effets possibles de ces produits. Chaque classe de contaminants chimiques identifiée pourrait faire l'objet d'un examen détaillé distinct. Il sera également question des pesticides, des médicaments, des polluants organiques persistants et des métaux dans le contexte de l'industrie aquacole canadienne.

Deux classes de produits ne nécessiteront pas davantage de recherche : les additifs alimentaires, notamment les antioxydants (agents de conservation) et les caroténoïdes (coloration de la chair), qui n'ont sans doute pas d'effets sur l'environnement. Quant au MS-222 (méthanesulfonate de tricaïne), utilisé par l'industrie aquacole du Nouveau-Brunswick, Zitko (1994) souligne qu'on ne prévoit aucun effet environnemental néfaste dû à son utilisation.

Les produits chimiques utilisés dans l'industrie aquacole canadienne sont présentés dans le tableau 1. Ce dernier résume les données scientifiques récentes sur l'utilisation, la persistance et les effets potentiels sur l'environnement de ces produits. Il y a relativement peu de publications primaires qui traitent du devenir et des effets des produits chimiques utilisés dans l'aquaculture au Canada. Il est évident qu'il existe un certain nombre de lacunes dans les connaissances sur chaque produit ou classe de produit. Un examen plus détaillé de chaque produit permettrait de cerner davantage de lacunes au sujet de celui-ci.

Il semble qu'il n'y ait aucune donnée publiée sur les antibiotiques présents à proximité des emplacements aquacoles du Canada, que ce soit sur leur présence dans les sédiments ou dans le biote aquatique, sur la présence et l'incidence d'organismes résistants aux antibiotiques dans les sédiments ou chez les espèces indigènes, ou sur les résidus d'antibiotiques présents chez les poissons et les organismes aquatiques non visés. L'accumulation d'antibiotiques dans les sédiments pourrait nuire aux communautés bactériennes et à la minéralisation des déchets organiques (Stewart, 1994), mais aucune étude sur le sujet n'a été publiée au Canada.

La majorité des travaux effectués jusqu'à maintenant sur les pesticides ont eu lieu en laboratoire et ont porté sur les effets immédiats des produits chimiques de lutte contre le pou du poisson sur les organismes aquatiques (espèces non visées). Le nombre limité d'essais menés sur le terrain ont porté sur la létalité à la suite de traitement ponctuel. Les effets à court terme de l'utilisation de pesticides et les études à long terme visant à déterminer la variabilité naturelle des populations locales et les mesures du changement de la biodiversité doivent être évalués. Jusqu'à maintenant, la majorité des études sur les effets des produits chimiques ont été menées sur les espèces non visées importantes sur le plan commercial. Il n'existe apparemment aucune donnée sur les effets de ces produits chimiques sur les micro-organismes et les espèces planctoniques qui constituent la base de la chaîne alimentaire marine dans les milieux côtiers. Les compositions chimiques des pesticides et des désinfectants n'ont pas été déterminées, et nombre de leurs ingrédients « inertes » peuvent être toxiques pour le biote aquatique (Zitko, 1994).

On en connaît peu sur le lien entre la pisciculture et les contaminants environnementaux, tels que les polluants organiques persistants (POP) et les métaux. Les aliments pour poissons peuvent constituer une source de contamination des poissons d'élevage. Il est nécessaire de connaître les ingrédients de chaque produit pour évaluer ses effets possibles avec précision. Les métaux présents à proximité d'emplacements piscicoles peuvent provenir des deux autres sources suivantes : le lessivage à partir des cages en métal et les peintures antisalissures. Les concentrations de composés chlorés (Hellou et al., 2000) et de métaux (Chou et al., 2002) sont plus élevées lorsque la concentration totale de carbone organique des sédiments est élevée. Les cages en bois munies de flotteurs en mousse de polystyrène peuvent être une source de contaminants plastiques (Zitko, 1994). Cependant, on en connaît peu sur les effets des plastiques sur les organismes aquatiques.

De plus, il est possible de cerner des lacunes générales en rapport avec la démarche scientifique et la méthodologie :

- *La recherche sur les produits chimiques est nécessaire dans toutes les régions où l'on pratique la pisciculture marine au Canada. Les recherches doivent être poursuivies au Nouveau-Brunswick, où les scientifiques peuvent tirer profit d'une vaste base de données existante et où les conditions sont les meilleures pour surveiller les tendances à long terme. La recherche doit également être accrue à Terre-Neuve, en Nouvelle-Écosse et en Colombie-Britannique, où il y en a eu peu jusqu'à maintenant.*

- *Les données sur la toxicité se résument aux résultats des essais de létalité effectués sur de courtes périodes (p. ex. 24, 48 et 96 heures). Davantage de travaux sont nécessaires pour déterminer les effets chroniques, létaux et sublétaux, et les effets de doses réalistes de produits chimiques sur les espèces indigènes.*

- *Bien que l'on possède des données de laboratoire sur de nombreux produits, il n'existe que très peu de données de terrain sur les effets des produits chimiques utilisés dans la pisciculture. Il est nécessaire d'effectuer des relevés et des expériences de terrain sur les effets à court terme de l'utilisation de produits chimiques, de même que des études à long terme visant à déterminer la variabilité naturelle des populations locales et à mesurer les changements de la biodiversité (et d'autres indicateurs de la santé de l'écosystème).*

- *Les essais sur la toxicité sont menés en laboratoire et portent sur une seule espèce et un seul produit à la fois. Il existe un grave manque de données sur les effets cumulatifs de l'exposition à des produits chimiques et sur la concentration et le devenir des produits dérivés de la pisciculture. On doit déterminer les effets cumulatifs des produits chimiques et les effets d'expositions multiples sur les organismes non visés.*

Tableau 1. Résumé des produits chimiques utilisés dans l'industrie aquacole canadienne**

Produit	Utilisation	Persistance dans les sédiments	Bioaccumulation	Effets possibles
Oxytétracycline	Antibiotique	Persiste pour de longues périodes qui varient selon des facteurs environnementaux (Björklund et *al.*, 1990; Samuelsen, 1994; Hektoen et *al.*, 1995; Capone et *al.*, 1996); Demi-vie de 419 jours dans des conditions stagnantes et anoxiques (Björklund et *al.*, 1990).	Absorption par les huîtres et les crabes en laboratoire ou à proximité des cages à saumons (MPO, 1997); Concentration dans les tissus du crabe commun supérieure aux limites de l'USFDA (Capone et *al.*, 1996)	Résistance à l'oxytétracycline possible chez des poissons, des organismes non visés ou des communautés bactériennes vivant à proximité d'emplacements piscicoles (Björklund et *al.*, 1991; Hansen et *al.*, 1993; Hirvelä-Koski et *al.*, 1994)
Tribrissen	Antibiotique	Demi-vie estimée à 90 jours à une profondeur de 6 ou 7 cm (Hektoen et *al.*, 1995)		
Romet 30	Antibiotique		Absorption par les huîtres (Jones, 1990; LeBris et *al.*, 1995; Capone et *al.*, 1996; Cross, données non publiées)	
Florfénicol	Antibiotique	Demi-vie estimée à 4,5 jours (Hektoen et *al.*, 1995)		
Téflubenzuron	Médicament dans les aliments pour poisson pour la lutte contre le pou du poisson	Solubilité de 19 $\mu g \cdot L^{-1}$ et log K_{oe}[a] de 4,3 indiquent une possibilité de persistance (Tomlin, 1997); Persistance de plus de 6 mois dans une zone située à moins de 100 m d'une cage de poissons traités (SEPA, 1999b)		Inhibiteur de la production de chitine; cas de mortalité de homards juvéniles (LCPE, 1999b); atténuation possible en effectuant une dépuration avant la mue (McHenery, 1997; LCPE, 1999b)
Benzoate d'émamectine	Médicament dans les aliments pour poissons pour la lutte contre le pou du poisson	Solubilité de 5,5 $mg \cdot L^{-1}$ et log K_{oe} de 5 indiquent une possibilité de persistance (SEPA, 1999b)	Délai d'attente de 25 jours avant la vente du saumon	Perturbateur du mouvement des ions chlorure (Roy et *al.*, 2000); létal pour le homard à une concentration de 735 $\mu g \cdot kg^{-1}$ de nourriture (Burridge et *al.*, 2002); provoque la mue du homard (Waddy et *al.*, 2000c)
Ivermectine	Médicament dans les aliments pour poissons; traitement non indiqué sur l'étiquette pour la lutte contre le pou du poisson	Solubilité de 4 $mg \cdot L^{-1}$ (Tomlin, 1997); pourrait persister jusqu'à 28 jours (Wislocki et *al.*, 1989; Roth et *al.*, 1993)	Délai d'attente de 180 jours avant la vente; s'accumule dans les tissus du homard en 10 jours (Burridge, Haya et Zitko, données non publiées)	Perturbateur du mouvement des ions chlorure (Roy et *al.*, 2000); mortalité cumulative de 80 % des saumons atlantiques exposés à 0,2 $mg \cdot kg^{-1}$ pendant 27 jours (Johnson et *al.*, 1993); CL_{50} 96 h = 8,5 $mg \cdot kg^{-1}$ de nourriture pour crevettes; CSEO[b] était de 2,6 $mg \cdot kg^{-1}$ de nourriture (Burridge et Haya, 1993)

Tableau 1 (suite). Résumé des produits chimiques utilisés dans l'industrie aquacole canadienne**

Produit	Utilisation	Persistance dans les sédiments	Bioaccumulation	Effets possibles
Azaméthiphos	Pesticide; traitement dans un bain pour la lutte contre le pou du poisson	Solubilité de 1,1 $\mu g \cdot L^{-1}$ et log K_{oe} de 1,05; ne devrait pas persister (Tomlin, 1997)	Accumulation dans les tissus improbable (Roth et *al.*, 1993 et 1996)	Neurotoxine, inhibiteur de l'acétylcholinestérase, mais ne s'accumule pas (Roth et *al.*, 1993 et 1996); mutagène *in vitro* (Comité des médicaments vétérinaires, 1999; Zitko, 2001); bain d'une heure à une concentration de 1 $mg \cdot L^{-1}$: létal pour 15 % des saumons après 24 h (Sievers et *al.*, 1995); CL_{50} 48 h = 3,57 – 1,39 $\mu g \cdot L^{-1}$ et CSEO 120 min = 1 $\mu g \cdot L^{-1}$ pour les larves et les adultes du homard (Burridge et *al.*, 1999a et 2000a); réactions comportementales à une concentration supérieure à 10 $\mu g \cdot L^{-1}$ (Burridge et *al.*, 2000a et b)
Peintures antisalissures à base de cuivre	Agent antisalissures; réduction des salissures sur les filets	Concentration élevée de cuivre (Cu) dans les sédiments (Burridge et *al.*, 1999a)	Accumulation possible dans le biote aquatique	Concentration de 100 à 150 $mg(Cu) \cdot kg^{-1}$ dans les sédiments peut nuire à la diversité de la faune benthique (Debourg et *al.*, 1993); dans la plupart des sites d'échantillonnage : concentration supérieure à la RPQS[c] de 18,7 $mg \cdot kg^{-1}$ et létale pour les amphipodes et les échinides (Burridge et al., 1999a)
Iodophores	Désinfectants d'équipement	Ne devrait pas persister (Zitko, 1994)		Préparations peuvent contenir des composés néfastes ou toxiques pour le biote aquatique (Zitko, 1994; Madsen et *al.*, 1997; Ashfield et *al.*, 1998)
Chlore/hypochlorite	Désinfectants; nettoyage de filets			Toxiques pour les organismes aquatiques (Zitko, 1994)
BPC, HAP, p,p'-DDE	Présents dans les aliments pour poissons (Zitko, 1994)	BPC indécelables pour une limite de détection = 0,05 - 0,10 $\mu g \cdot g^{-1}$ en poids sec (Burridge et *al.*, 1999a); p,p '-DDE décelable pour une limite de détection = 1 $ng \cdot g^{-1}$ en poids sec (Hellou et *al.*, 2000)	Modification du profil lipidique des poissons sauvages (Zitko, 1994)	

Tableau 1 (suite). Résumé des produits chimiques utilisés dans l'industrie aquacole canadienne**

Produit	Utilisation	Persistance dans les sédiments	Bioaccumulation	Effets possibles
Cadmium, plomb, cuivre, zinc et mercure	Matériaux des cages; aliments pour poissons	Concentrations de cuivre (>2 fois) et de zinc (de 1 à 2 fois) plus élevées dans les sédiments sous les cages que dans les aliments pour poissons (Chou et *al.*, 2002); Concentration de cadmium dépasse 0,7 $\mu g \cdot g^{-1}$ (Burridge et *al.*, 1999a)	Toxicité ou accumulation possible dans le biote aquatique	
Billes de polystyrène	Flotteurs en mousse de polystyrène	Source de contaminants à faible poids moléculaire (Zitko, 1994)		Changement de la faune benthique par modification des échanges gazeux dans les eaux interstitielles, par ingestion ou par création d'habitats pour des organismes opportunistes (Goldberg, 1997).

** Le tableau ne comprend que les composés utilisés au Canada (actuellement ou dans le passé). D'autres classes de composés sont utilisées couramment ailleurs et elles pourraient un jour être disponibles au Canada.

a – log K_{oe} = logarithme du coefficient de partage octanol/eau. Il est internationalement reconnu qu'un log K_{oe} >= 3 représente une possibilité de bioaccumulation. D'après la *Loi canadienne sur la protection de l'environnement (LCPE)*, un log K_{oe} >= 5 représente une possibilité de persistance ou de bioaccumulation (Beek et *al.*, 2000).

b – CSEO = concentration sans effet observé

c – RPQS = Recommandations provisoires pour la qualité des sédiments

INTRODUCTION

Salmonid aquaculture is a relatively new industry in Canada. Commercial operations began in the late 1970s. British Columbia and New Brunswick are the largest producers of Atlantic salmon (*Salmo salar)* in Canada (Canadian Aquaculture Industry Alliance 2000). Other provinces in which marine finfish aquaculture is practiced are as follows. Newfoundland's finfish industry is based primarily on the culture of rainbow/steelhead trout (*Oncorhynchus mykiss)* and Atlantic salmon reared in sea cages. Prince Edward Island's main finfish production is found in the culture of Arctic charr (*Salvelinus alpinus*) and rainbow trout. The Nova Scotia finfish industry produces mainly rainbow/steelhead trout in land-based facilities and marine cage sites. New Brunswick's primary finfish species is Atlantic salmon. The Quebec industry is primarily focused on producing rainbow trout for the food market and speckled trout (*Salvelinus fontinalis*) for enhancement of the sport-fish trade. There are no cage culture operations in Quebec. Plans are being made for placing cages in the Baie de Gaspé in the near future (M. Patterson, Sociéte de développement de l'industries maricole, Gaspé, QC, personal communication). British Columbia is the largest producer of Atlantic salmon in Canada. Coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*Oncorhynchus tshawytscha*) are also grown commercially in British Columbia.

Beginning in 1994, the east coast aquaculture industry has dealt with a succession of serious parasite and disease problems. As early as 1993, sea lice were identified as a major concern for the aquaculture industry in British Columbia (Constantine 1994). Management practices were instituted that have resulted in greatly improved husbandry and a reduction in the use of some chemicals, particularly antibiotics. However, fish farmers continue to use chemotherapeutants to treat infestations of ectoparasites, as well as disinfectants to manage spread of diseases.

Chemotherapeutants used in the aquaculture industry are considered either a drug or a pesticide depending on the use and method of application. Health Canada regulates the use of both drugs and pesticides and makes the distinction between them based on the method of application. If the product is applied topically or directly into water, it is considered a pesticide; however, if a product is delivered through medicated feed or by injection, it is considered a drug.

Drug approval is the responsibility of the Veterinary Drugs Directorate (VDD) of Health Canada, under the authority of the *Food and Drugs Act* (FDA). The FDA does not require the submission of environmental data; however, the mandate of the VDD is to ensure that drugs sold for use in animals are safe and effective and do not leave residues in food animal products, which could present a human health risk to the consumer. There are provisions for Emergency Drug Release (EDR) and 'off-label' use of a drug. The emergency release of a drug must have VDD authorization before a manufacturer is allowed to sell an unapproved new drug that does not have a Drug Identification Number (DIN) to a licensed veterinarian for the emergency treatment of (a) patient(s). It is preferred that the 'off-label' use of any drug with a DIN is avoided whenever an approved product is available. Veterinarians who use drugs in an 'off-label' manner are responsible for animal safety and for any illegal drug residues that are detected in animal products sold for human consumption (Salmon Health Consortium 2002).

Pesticides are the responsibility of the Pest Management Regulatory Agency (PMRA) of Health Canada and are registered under the authority of the *Pest Control Products Act* (PCPA). Herbicides, fungicides, disinfectants, insecticides and antimicrobials are all considered pesticides. Antifouling agents that are added to paints and coatings to restrict growth of aquatic fouling pest organisms, such as algae and molluscs, on net-pens are also registered under the PCPA. The PCPA requires the registrant to submit environmental data as part of the registration process. Most data submitted to the regulatory agencies are proprietary and, as such, are not available to the general public but can be obtained by researchers (with restrictions) from Health Canada.

Burka et al. (1997) summarized the drugs used in aquaculture. However, with the exception of some recent publications regarding the lethality of chemotherapeutants to American lobsters (*Homarus americanus*) (Burridge et al. 2000a,b,c), there are few publications in the primary literature regarding the environmental fate and effects of these drugs, other chemotherapeutants and chemicals in general used in the Canadian aquaculture industry.

There has been a great deal of scientific debate regarding the environmental consequences of chemical usage in aquaculture. The debate has also moved into the public domain: where views of the opposing sides are typified by several highly publicized anti-aquaculture articles and, most recently, television documentaries (Ellis 1996; Goldburg and Triplett 1997; Milewski et al. 1997), and the responses to these articles from the finfish aquaculture industry (e.g. The Canadian Aquaculture Industry Alliance 2001a,b).

Reviews of the potential consequences of chemical use in aquaculture or introduction from aquaculture activities include those prepared by Zitko (1994) and GESAMP (1997). Issues raised and recommendations made by these authors have yet to be addressed in a significant manner. In addition, the authors of recent reviews of environmental impacts of aquaculture have identified chemical inputs from aquaculture activity as an area requiring further research (Nash 2001; Anonymous 2002).

Several projects recently funded by Fisheries and Oceans Canada’s (DFO) Environmental Science Strategic Research Fund (ESSRF) have allowed scientists to begin to address some of these topics. However, researchers are still in the early stages of identifying sources of contamination and potential effects on the environment, particularly to non-target species.

This manuscript is a summary of potential sources of chemical contamination, chemicals which may be involved and knowledge about the effects of these compounds. This review cannot be considered comprehensive. Each of the various classes of chemicals identified could be the subject of a comprehensive review. The author has limited discussion to chemicals that are relevant to the Canadian aquaculture industry. The author identified compounds for discussion using the general classification of aquaculture chemicals described by Zitko (1994) and Haya et al. (2001). These authors suggest organic and inorganic compounds may be introduced to the marine environment as the result of aquaculture activity, either intentionally or unintentionally, from various sources including medicinals, feed constituents and construction materials. The discussion of medicinal products is limited to compounds that are registered by Health Canada.

Pesticide use in marine finfish aquaculture has been limited to treatment of salmon against sea lice infestations. Infestations of sea lice are prevalent in New Brunswick, Nova Scotia and British Columbia. Incidents of sea lice have not been reported in Newfoundland (NF) (G. Perry, Northwest Atlantic Fisheries Centre, St. John's, NF, personal communication). Recently, there has been a suggestion that bath treatments using formaldehyde may be taking place to treat parasites present on gills of Atlantic salmon in southwest New Brunswick (NB) (B.D. Chang, St. Andrews Biological Station, St. Andrews, NB, personal communication).

Compounds which have been applied as anti-sea lice agents in Canada are as follows: azamethiphos, ivermectin, emamectin benzoate and teflubenzuron. Azamethiphos is applied as a bath treatment and is considered a pesticide by Health Canada. It is fully registered after review by PMRA. The latter three compounds are in-feed additives and are thus considered drugs by Health Canada. Their use is regulated by Health Canada's Veterinary Drugs Directorate. Ivermectin is used under veterinary prescription as an 'off-label' product. Emamectin benzoate and teflubenzuron are used under Emergency Drug Release from Health Canada.

AZAMETHIPHOS

Azamethiphos is an organophosphate insecticide and the active ingredient in the formulation Salmosan®. This is the only pesticide currently registered by Health Canada (PMRA) to treat salmon against infestations of sea lice in Canada (Health Canada 2002). It is used as a bath treatment of 1 h at 100 $\mu g \cdot L^{-1}$ when water temperatures are less than 10ºC and for 30 min at 100 $\mu g \cdot L^{-1}$ when water temperatures are greater than 10ºC. Azamethiphos has a high water solubility (1.1 $g \cdot L^{-1}$) and a low octanol-water partition coefficient (log K_{ow} = 1.05) (Tomlin 1997). This characteristic provides an indication of a chemical's likely partitioning between water and sediment or biota. However, predictions regarding persistence require data regarding physical and chemical degradation. Azamethiphos is unlikely to accumulate in tissue or in sediment. Azamethiphos has neuro-toxic action, acting as an acetylcholinesterase (AChE) inhibitor. The depression of AChE by azamethiphos is not cumulative (Roth et al. 1993, 1996). Azamethiphos has been shown to be mutagenic in several *in vitro* tests (Committee for Veterinary Medicinal Products 1999).

Recently, Zitko (2001) reported the alkylating potency of azamethiphos to be fairly high. He suggested this property may explain the mutagenicity of azamethiphos and indicated that future studies with this compound and non-target biota should include tests for delayed effects (Zitko 2001).

A 1-h bath in azamethiphos at 1 mg·L^{-1} resulted in 15% mortality of Atlantic salmon after 24 h (Sievers et al. 1995). Roth et al. (1993) reported that herring larvae tolerated azamethiphos better than another anti-lice chemical, dichlorvos. This is the only publication found that reported consequences of use of this compound to non-target fish.

Results of lethality studies with azamethiphos show lobster and shrimp are the most sensitive of the species tested. Bivalves such as scallops and clams were unaffected (Burridge and Haya 1998). Adult lobsters held within the tarpaulin during an operational treatment did not survive. The 48-h LC50 has been estimated for the first four larval stages of the American lobster and adults of the same species (Burridge et al. 1999b). The values are as follows: Stage I 3.57 µg·L^{-1}, Stage II 1.03 µg·L^{-1}, Stage III 2.29 µg·L^{-1}, Stage IV 2.12 µg·L^{-1}, and Adults 1.39 µg·L^{-1}. There is no statistically significant difference between these values. Adult and Stage IV lobsters were exposed repeatedly (up to nine times) for varying lengths of time to four concentrations of azamethiphos (Burridge et al. 2000a). The No Observed Effect Level (NOEL) was repeated at 120-min exposures to 1 µg·L^{-1} of azamethiphos. In addition to observed lethality, several surviving lobsters showed significant behavioral responses to repeated exposure to concentrations greater than 10 µg·L^{-1} (Burridge et al. 2000a).

Lobsters exposed to azamethiphos became quite agitated, often 'flopping' erratically around the exposure tank. They were also aggressive to other lobsters and reacted very quickly to any movement. They seemed to lose control of their claws and eventually flipped onto their backs and died within hours (Burridge et al. 2000a). Affected survivors may remain moribund for periods of time ranging from hours to days. The consequences of behavioral responses such as these on organisms and populations in the natural environment are unknown.

Laboratory studies have been conducted to investigate possible sublethal effects of azamethiphos exposure on the American lobster. Preovigerous females were exposed for 1 h biweekly to 10 µg·L^{-1} azamethiphos and monitored for spawning success and survival. Surprisingly, even with such infrequent exposures, up to 100% of the animals exposed to this concentration died during the experiment: some expired after only three treatments. Significant numbers of the surviving lobsters failed to spawn (Burridge et al. 2000b; Waddy et al. 2002a). Roth et al. (1993, 1996) state that the inhibition of AChE by azamethiphos is not cumulative in fish. There does, however, appear to be a cumulative inhibition of AChE in lobster (L. Burridge, unpublished data). Work is underway by these researchers to determine if a dose-response relationship exists between azamethiphos and this endpoint.

Abgrall et al. (2000) investigated the effects of azamethiphos on lobster behavior. They found that shelter use could be affected by azamethiphos. However, exposure to water concentrations greater than five times the recommended treatment concentration for periods of several hours was necessary.

During 1995, a field trial was conducted to determine the effects of single operational azamethiphos treatments on juvenile and adult lobsters, shrimp, clams and scallops.

Animals were suspended at two depths and varying distances from the treated cage. During two trials, lobsters held within the treatment tarp died. No other treatment-related mortalities were observed (Chang and McClelland 1996). In addition, lobsters were suspended at three depths at 20 sites surrounding a salmon cage site that was conducting operational treatments with azamethiphos. No treatment-related mortalities were observed (Chang and McClelland 1997).

Finally, survival of lobsters suspended at mid-depth and near bottom at four sites in the salmon farming area of Lime Kiln Bay, plus a control site, was monitored for nine weeks during August-October 1996. There were no apparent differences in lobster survival between the experimental and control sites (Chang and McClelland 1997). No residues of azamethiphos were detected in water samples collected weekly from the five sites (Detection Limit = 50 $pg{\cdot}L^{-1}$) (Chang and McClelland 1997).

Diving surveys were conducted at a lobster nursery area located near a salmon farm in early August, September and late October of 1996. There were no apparent changes in lobster populations over time, and the area was found to have a considerable population of juvenile lobsters (Chang and McClelland 1997).

Measurements of primary productivity and dissolved oxygen were made before, during and after chemical treatments at salmon farms in southwest New Brunswick in August-September 1996. There were no evident effects on dissolved oxygen and chlorophyll *a* levels, indicating no impact on primary production (D. Wildish, St. Andrews Biological Station, St. Andrews, NB, unpublished data).

AVERMECTINS

Two compounds from the avermectin class of drugs have been used to treat sea lice infestations. Ivermectin and emamectin benzoate are semi-synthetic derivatives of a chemical produced by the bacterium, *Streptomyces avermitilis*. They are effective in the control of internal and external parasites in a wide range of host species, particularly mammals (Campbell et al. 1983). The avermectins generally open glutamate-gated chloride channels. The result is an increase in chloride concentrations, hyperpolarization of muscle and nerve tissue, and inhibition of neural transmission (Roy et al. 2000; Grant 2002).

Ivermectin has a low water solubility of 4 $mg{\cdot}L^{-1}$ (Tomlin 1997). It has been suggested that ivermectin will persist in sediments for more than 28 days (Wislocki et al. 1989; Roth et al. 1993). It is applied as an 'in-feed' additive at doses ranging from 0.05 to 0.2 $mg{\cdot}kg^{-1}$ of fish. Its safety margin to fish is quite small (see below). Fish are therefore fed treated food twice weekly for two weeks. There is some discussion as to whether ivermectin persists in fish tissue (Roth 1993). However, a 180-day withdrawal period is required prior to marketing the treated fish. Ivermectin was used to treat sea lice as an 'off-label' drug treatment under veterinary prescription. This means the drug (and product) has regulatory status from Health Canada but is not labelled for the specific treatment – in this case anti-sea lice treatment. As a drug, ivermectin has not been subject to the scrutiny of the registration procedure that other sea lice pesticides have been.

Johnson et al. (1993) reported a cumulative mortality of 10% and 80% of Atlantic salmon (wt = 800 g) exposed to 0.05 and 0.2 $mg{\cdot}kg^{-1}$ ivermectin (in food), respectively, over a 27-day period. Atlantic salmon was the most sensitive of several salmonid species tested. These authors also reported behavioral changes, such as cessation of feeding and lethargy, in fish exposed to lower concentrations.

Roth et al. (1993) briefly reviewed the use of ivermectin to treat sea lice. These authors stated that the available data on the toxicity of ivermectin to non-target marine organisms were very limited. They also cautioned that ivermectin's tendency to bind tightly to sediment and the lack of information regarding its effects on benthic organisms present significant drawbacks to its use in sea lice control.

Burridge and Haya (1993) reported results of an experiment in which sand shrimp (*Crangon septemspinosa*) were exposed to food treated with various concentrations of ivermectin for 96 h in running seawater. When the food was accessible by the shrimp, mortality occurred. When the food was present in the water but not accessible by the shrimp, no mortality occurred, suggesting the food must be ingested by the shrimp before lethality occurs. The nominal 96-h LC50 was 8.5 $mg{\cdot}kg^{-1}$ food. The No Observed Effect Concentration (NOEC) was 2.6 $mg{\cdot}kg^{-1}$ food.

Tests have also been conducted to determine the lethality of ivermectin to two species of crabs (green and rock) and to adult lobsters. Small numbers of animals were exposed to ivermectin treated food for several days and weeks. No animals died during any exposures. Ivermectin levels in lobster tissue (meat and hepatopancreas) were measured. Ivermectin accumulated in these tissues over a 10-day period. More ivermectin was found in the hepatopancreas than in the muscle. These experiments were preliminary (i.e. with low sample numbers and no replicates) (L. Burridge, K. Haya and V. Zitko, St. Andrews Biological Station, St. Andrews, NB, unpublished data). If ivermectin use continues, this study should be repeated in a more rigorous fashion to establish the bioaccumulation potential of ivermectin in lobsters.

With the availability of emamectin benzoate as a treatment against sea lice infestations (see below), the use of ivermectin has been greatly reduced. In fact, the presence of an alternative treatment should eliminate the need for the use of 'off-label' prescriptions.

Emamectin benzoate (EB) is the benzoate salt of emamectin, another semi-synthetic avermectin. It is the active ingredient in the in-feed sea lice treatment Slice® (Roy et al. 2000). EB has a low seawater solubility (5.5 $mg{\cdot}L^{-1}$) and an octanol/water partition coefficient (log K_{ow}) of 5, suggesting the compound has a potential to be absorbed and bound to particulate material and surfaces (SEPA 1999a). In fact, studies showed that emamectin benzoate reaching soils and sediments was tightly bound (SEPA 1999a). The optimum therapeutic dose is 50 $\mu g{\cdot}kg^{-1}$ (fish) for seven consecutive days (Stone et al. 1999). Toxicological studies have shown that emamectin benzoate is less toxic than ivermectin in all taxa where tests were conducted using each compound (SEPA 1999a). Van den Heuval et al. (1996) exposed bluegill sunfish to avermectin B_{1a} (a compound similar to emamectin benzoate) and measured bioconcentration factors. The authors concluded that, despite the relatively high log K_{ow}, these compounds will not bioaccumulate in aquatic organisms.

Slice® has been available as an EDR from Health Canada since 1999 and is used to treat salmon against sea lice in eastern Canada. Emamectin benzoate is active against all life stages of the sea louse, making it more effective than azamethiphos (Stone et al. 1999, 2000). In-feed treatments also tend to be less labor intensive and less stressful to the fish compared to bath treatments. The withdrawal period prior to slaughter of salmon in Canada is 25 days, much shorter than that of ivermectin (180 days).

Roy et al. (2000) reported that exposure of Atlantic salmon and rainbow trout to emamectin benzoate at concentrations up to ten times the recommended treatment concentration resulted in no mortality. Signs of toxicity were only observed at the highest treatment concentration (Roy et al. 2000).

The lethality of emamectin benzoate (in treated fish feed) to American lobsters is estimated to be 735 $\mu g \cdot kg^{-1}$ (of feed) (Burridge et al. 2002; Waddy et al. 2002b). The treatment concentration ranges from 1 to 25 $\mu g \cdot kg^{-1}$ (Roy et al. 2000). In a recent publication, Waddy et al. (2002c) reported that ingestion of emamectin benzoate by lobster can lead to premature molting. Although work is ongoing, it appears as though the concentration necessary to produce this sublethal response is higher than normally used in field situations (S. Waddy, St. Andrews Biological Station, St. Andrews, NB, personal communication).

TEFLUBENZURON

Teflubenzuron is the active ingredient in the in-feed sea lice treatment Calicide® (SEPA 1999b). The product acts by inhibiting the formation of chitin, the predominant component in the exoskeleton of crustacea. As such, the product is efficacious only against molting sea lice (Branson et al. 2000). The recommended treatment regime is 10 mg (teflubenzuron) per kg of salmon per day for seven days (SEPA 1999b). Teflubenzuron is sparingly soluble in water (19 $\mu g \cdot L^{-1}$) and has a log K_{ow} of 4.3, indicating it has a potential to be absorbed and bound to particulate and organic material and sediments (Tomlin 1997; SEPA 1999b). In studies conducted in Europe, teflubenzuron was found to persist in sediment for longer than six months, but the extent of contamination was limited to an area less than 100 m from the treated cage (SEPA 1999b). In addition, researchers found treatment-related mortalities in juvenile lobster suspended near treated cages (SEPA 1999b). SEPA concluded that there was a risk to sediment dwelling crustacea; although this may be mitigated by depuration of the compound by these animals prior to molting (McHenery 1997; SEPA 1999b). The product received an EDR from Health Canada in 1998.

There is increasing concern among aquaculturists that sea lice will develop widespread resistance to chemotherapeutants (Denholm et al. 2002). Sea lice have been shown to become resistant to treatment with organophosphates and pyrethroids in Europe (Denholm et al. 2002). With limited treatment options available, resistance development could result in more frequent treatment and an increase in the quantity of chemotherapeutants entering the marine environment.

PERSISTENT ORGANICS

Little is known about contaminants (persistent organic pollutants (POPs) and metals) present in fish feed. Fish oils (herring, shark), meals (fish, wheat, blood, poultry, canola, corn gluten), essential minerals, dyes and antioxidants are constituents of fish feed (Zitko 1994). Until recently, many of the fish oils used in preparation of fish feed may have contained polychlorinated biphenyls (PCBs) (Zitko 1994).

In addition, Zitko (1994) reported that the lipid profiles of local wild fish populations are changing, possibly as a consequence of the presence of aquaculture feed. Currently, most of the fish meals and oils used in feed production come from South America and are less likely to be contaminated with PCBs (D. Higgs, West Vancouver Laboratory, Vancouver, BC, personal communication).

The author is aware of only two studies that have investigated the presence and concentration of organic compounds near aquaculture sites. Burridge et al. (1999a) reported no detectable concentrations of mono- through tetra-chlorophenols in sediment samples collected near aquaculture sites. Similarly, PCBs were not detectable in these samples (Detection Limit = 0.05-0.1 $\mu g \cdot g^{-1}$ dry weight).

Hellou et al. (2000) investigated the presence of chlorinated pesticides, PCBs and polycyclic aromatic hydrocarbons (PAHs) in sediments collected near aquaculture sites in southwest New Brunswick. Five alkylated PAHs were present and accumulated in sediments. Levels of total PAHs were lower in sediments collected directly under cages relative to 25 m away. These data suggest a presence of PAHs that originated from sources other than aquaculture, perhaps combustion sources. The increased organic loading near aquaculture sites actually reduces the apparent concentration of PAHs. Of twelve chlorinated pesticides measured, only p,p'-DDE was consistently detected in sediments (Detection Limit = 1 $ng \cdot g^{-1}$ dry weight). Of the 159 PCB congeners, mainly 153/168/132 (unresolved) were detected in these same samples (Detection Limit = 1 $ng \cdot g^{-1}$ dry weight). These chlorinated compounds were present at very low concentrations and observed to be higher where total organic carbon content was higher (i.e. under the cages).

The Province of New Brunswick has implemented an environmental monitoring plan (EMP) whereby sites are rated according to sediment quality using several subjective as well as geochemical criteria (Hargrave et al. 1993). Organochlorines were lowest at sites given an EMP rating of A (normoxic), intermediate at sites rated B (hypoxic) and highest at sites rated C (anoxic).

Easton et al. (2002) measured PAHs in commercial fish feed and farmed and wild salmonids from British Columbia. This was a preliminary investigation with only a small number of samples being analyzed and no replication of sampling. They found the highest concentrations of these compounds in a single wild chinook followed by commercial feed.

In addition to work with sediments, Hellou et al. (2000) also investigated the presence of chlorinated pesticides, PCBs and PAHs in fish feed. The same five alkylated PAHs identified in sediments were present in feed pellets and fish oil. Two other alkylated PAHs as well as three parental PAHs were in feed, but not accounted for in fish oil, while other ingredients used to prepare feed were not analyzed for contaminants. As was the case with sediments, only p,p -DDE was consistently detected in pellets and fish oil (Detection Limit = 1 $ng \cdot g^{-1}$ dry weight). Of the 159 PCB congeners, mainly 153/168/132 (unresolved) were detected at very low concentrations in fish feed and fish oil (Detection Limit = 1 $ng \cdot g^{-1}$ dry weight).

Analyses have also been performed on fish feed blends from Canada's west coast (M. Ikonomou, Institute of Oceans Sciences, Sidney, BC, personal communication). Dioxins and furans ranged in concentration from 8 to 25 $ng \cdot kg^{-1}$, the full range of PCB congeners from 35 to 55 $\mu g \cdot kg^{-1}$, 25 organochlorine pesticides from 25 to 65 $\mu g \cdot kg^{-1}$, and toxaphene from 40 to 55 $\mu g \cdot kg^{-1}$ (M. Ikonomou, Institute of Oceans Sciences, Sidney, BC, unpublished data). Easton et al. (2002) reported similar results for fish feed from the west coast of Canada (total organochlorine pesticides from 25 to 63 $\mu g \cdot kg^{-1}$, toxaphene from 10 to 55 $\mu g \cdot kg^{-1}$ and dioxin-like PCBs from 3 to 10 $ng \cdot kg^{-1}$).

The Canadian Food Inspection Agency (CFIA) (2002) has also reported levels of dioxins and furans, PCBs and DDT in fish meal, fish feed and fish oil of various origins. They reported mean dioxin and furan levels ranging from 0 to 1.1 $ng \cdot kg^{-1}$ toxic equivalent values (TEQ) for fish meal and fish feed and 3.7 to 9.9 $ng \cdot kg^{-1}$ TEQ for fish oil. These values are well below regulatory guidelines: Canadian Guidelines for Chemical Contaminants and Toxins in Fish and Fish Products (maximum limit of 20 $ng \cdot kg^{-1}$ TEQ) (CFIA 2002). The mean concentration of total PCBs in fish meal and fish feed ranged from 0.6 to 30.7 $\mu g \cdot kg^{-1}$. Mean concentrations of PCBs in fish oil ranged from 130.7 to 271.4 $\mu g \cdot kg^{-1}$. When these data are converted to TEQ for PCBs (all values <5 $\mu g \cdot kg^{-1}$), they fall well below the maximum limit prescribed by the Canadian Guidelines for Chemical Contaminants and Toxins in Fish and Fish Products of 2 $mg \cdot kg^{-1}$ (CFIA 2002). The mean concentration of DDT in fish meal and fish feed ranged from non-detectable to 23.3 $\mu g \cdot kg^{-1}$ and ranged from 7 to 90 $\mu g \cdot kg^{-1}$ in fish oil. Again, these concentrations fall well below the maximum limit prescribed by the Canadian Guidelines for Chemical Contaminants and Toxins in Fish and Fish Products of 0.5 $mg \cdot kg^{-1}$ (CFIA 2002). These data also confirm that dioxin and furan as well as PCB concentrations are lower in fish meal, fish feed and fish oil originating in South America than in those from North America and Europe.

Additional food additives include the carotenoids, canthaxanthin and astaxanthin, which are added to food to get a good flesh color in the salmon prior to marketing. Burridge et al. (1999a) identified canthaxanthin in sediment samples collected directly under salmon cages. Zitko (1994) described these compounds and suggested they are unlikely to affect non-target organisms. Antioxidants such as ethoxyquin, butylated hydroxyanisole (BHA) and butylated hydroxytoluene (BHT) may be added to food to prolong its shelf-life (Zitko 1994; D. Higgs, West Vancouver Laboratory, Vancouver, BC, personal communication). No data are available regarding effects of these compounds on non-target organisms or on their fate near aquaculture sites.

ANTIMICROBIALS

Health Canada (2001) identifies oxytetracycline, sulfadiazine 20% and trimethoprim 80%, sulfadimethoxine and ormetoprim (5:1), and florfenicol as drugs specifically registered for use in aquaculture. Possible areas of research related to the use of antibiotics in aquaculture include the following: persistence of the compound(s); residues and effects in non-target organisms; development of resistance; and promotion of antibiotic-resistant strains of microorganisms.

Several authors have shown oxytetracycline to persist in sediment near aquaculture cages for long periods of time, depending on sedimentation rates, water temperature and other environmental factors (Björklund et al. 1990; Samuelsen 1994; Hektoen et al. 1995; Capone et al. 1996). The half-life of oxytetracycline in sediment can be as long as 419 days under stagnant, anoxic conditions (Björklund et al. 1990). The half-life of Tribrissen (a formulation of sulfadiazine 20% and trimethoprim 80%) was estimated to be as long as 90 days at sediment depths of 6 to 7 cm (Hektoen et al. 1995). Florfenicol concentrations decreased rapidly in sediments with an estimated half-life of 4.5 days (Hektoen et al. 1995).

In the Report to the Provincial Environmental Assessment Review of Salmon Aquaculture in British Columbia (DFO 1997), reference is made to uptake of oxytetracycline by oysters and crabs and Romet 30 (a formulation of sulfadimethoxine and ormetoprim (5:1)) by oysters (Jones 1990; LeBris et al. 1995; Capone et al. 1996; Cross unpublished data). Each study showed at least some uptake of the antibiotic by these invertebrates either in the laboratory or in close proximity to salmon cage sites. The concentration of oxytetracycline in rock crab was as high as 3.8 $\mu g \cdot g^{-1}$, well in excess of the US Food and Drug Administration limit for commercially sold seafood (2 $\mu g \cdot g^{-1}$) (Capone et al. 1996). There appear to be no published reports of detectable antibiotic residues in fish and invertebrates from Canadian aquaculture sites. There is also a lack of information regarding the effects of antibiotic contamination in non-target species.

Stewart (1994) suggested that accumulation of antibiotics in sediments may interfere with bacterial communities and affect mineralization of organic wastes. In intensive aquaculture, antibiotics are used universally to treat diseases and, in the past, there was widespread prophylactic use. Antibiotics may reach the environment and lead to the selection of resistance in non-target benthic organisms (GESAMP 1997).

Resistance to antibiotics may occur in fish, non-target organisms and the bacterial community present in sediments near aquaculture activities. Hansen et al. (1993) reported an increase in antibiotic-resistant bacteria in sediments within a few days of onset of treatment with oxytetracycline or oxolinic acid. The presence of oxytetracycline resulted in higher numbers of antibiotic-resistant bacteria and a longer-lasting effect relative to other antibiotics and to control sites. Björklund et al. (1991) showed the presence of antibiotic-resistant bacteria in oxytetracycline-treated rainbow trout. Similarly, Hirvelä-Koski et al. (1994) identified antibiotic resistant strains of *Aeromonas salmonicida* in salmon at 9 of 35 fish farms treated with oxytetracycline. No resistance to other antibiotics was observed. Kerry et al. (1996) suggested caution should be exercised when interpreting results of experiments dealing with induction of antibiotic resistance. These authors identified the rate of cell division, as well as factors that affect cell division, as mitigating factors in establishing the validity of some studies. In addition, Smith et al. (1995) analyzed species frequency data and concluded operation of fish farms had only a negligible long-term impact on the microflora in under-cage sediments. Rhodes et al. (2000) examined the distribution of oxytetracycline resistant aeromonads in hospital and aquaculture environments in England. They found that inputs from human and aquaculture sources act in the same manner in terms of dissemination of tetracycline resistance-encoding plasmids. These data suggest examination of antibiotic resistance near aquaculture sites must take into consideration other possible sources of input.

It is anticipated there will be a continued reduction in antibiotic use in the Canadian finfish aquaculture industry (Sheppard 2000). In 2000, there were no prescriptions written for antibiotic use in Newfoundland (G. Perry, Northwest Atlantic Fisheries Centre, St. John's, NF, personal communication). Development of more effective vaccines and improved codes of practice have helped reduce the need for widespread use and eliminated prophylactic treatments. Nonetheless, some research is required to investigate the fate and effects of antibiotics in Canadian waters.

There appear to be no published data regarding the presence of antibiotics (in sediments and biota) of aquaculture origin in Canada. It is known that studies have been conducted and are ongoing. The author is aware of work by Cross et al. (1997), Zitko (St. Andrews Biological Station, St. Andrews, NB, unpublished data), Haya (St. Andrews Biological Station, St. Andrews, NB, unpublished data) and Hargrave et al. (Bedford Institute of Oceanography, Dartmouth, NS, unpublished data). It is unlikely that antibiotics such as oxytetracycline will behave in a significantly different manner in Canadian waters than in European waters. The prevalence of cold water temperatures in the Bay of Fundy, however, will likely result in oxytetracycline persisting for considerable lengths of time. Effects on indigenous non-target species have not been investigated. This may be important, given recent discussions regarding the institution of polyculture practices at some sites (K. Haya, St. Andrews Biological Station, St. Andrews, NB, personal communication). More research is required to understand if the use of antibiotics and other chemotherapeutants with specific target organisms have any unexpected and/or unwanted consequences for non-target or co-cultured species. Further information on antimicrobial use in marine finfish aquaculture would help develop research projects to address the remaining questions about presence and persistence of antibiotics in Canadian waters. The presence, prevalence and relevance of antibiotic resistant organisms in sediments and indigenous species around aquaculture sites must be investigated as well. These data can then be put in context by comparison to presence of antibiotic resistant bacteria in other aquatic environments (e.g. near sewage treatment plants).

ANTIFOULANTS

Copper-based antifouling paints are used to treat nets in aquaculture operations. Copper reduces the buildup of biota on nets which, in turn, allows the free flow of water and reduces the need for frequent net changes (Debourg et al. 1993). Copper (Cu) is highly toxic to aquatic organisms, may bioaccumulate, and concentrations greater than 100 to 150 $mg(Cu) \cdot kg^{-1}$ (dry weight) in sediment may reduce the diversity of benthic fauna (Debourg et al. 1993). A report submitted to the Scottish Executive (Anonymous 2002) suggests the use of copper antifoulant paints in aquaculture may be reason for concern due to its potential to accumulate in sediment.

The behavior and fate of copper from antifouling paints is dependent on factors such as pH, salinity and concentrations of organic and inorganic particles in water. Recently, Burridge et al. (1999a) reported elevated concentrations of copper in sediments near active aquaculture sites (relative to non-aquaculture sites) in southwest New Brunswick. At all but a few sample locations, the concentration of copper was higher than the Interim Sediment Quality Guideline (18.7 $mg \cdot kg^{-1}$) (Canadian Council of Ministers of Environment (CCME) 2002). These authors suggested that the elevated copper may have played a role in observed lethality in amphipod and echinoid fertilization bioassays as well as Microtox® tests (Burridge et al. 1999a). Parker and Aubé (2002) identified copper as being elevated relative to CCME guidelines in 80% of the aquaculture sites they sampled. The source of copper is unknown. While copper-based antifouling paint is also used to treat fishing, aquaculture support and recreation vessels which frequent the areas where samples were collected, these authors identified copper-treated nets as the most likely source of copper. Chou et al. (2002) also reported elevated copper concentrations in sediments relative to reference sites and related the copper concentration to EMP ratings. A site rated as hypoxic or anoxic is likely to have elevated concentrations of copper. These authors identifed fish feed as another source of copper.

In a recent study from Norway, Borufsen Solberg et al. (2002) found no difference in copper concentrations in tissues collected from fish, invertebrates, and macro-algae in and around aquaculture sites using copper-treated nets compared to samples collected from sites where copper-treated nets were not in use. Similarily, Lewis and Metaxas (1991) determined copper concentrations in water collected inside and outside a freshly (copper) treated salmon cage as well as 700 m away from the cage. These concentrations were not significantly different at the three sites and remained stable for over a month.

Historically, tri-butyl tin was used as an antifoulant. There is a considerable body of literature about the use and effects of tri-butyl tin on non-target organisms. The use of this active ingredient as a net coating has been banned and, to the author’s knowledge, none is being used in the aquaculture industry in Canada.

METALS

In addition to copper and tin from antifoulants, metals may be deposited near aquaculture sites from at least two other sources. Cages manufactured from metal (with plastic floats) and those made of plastics are the norm in the aquaculture industry today. Metals leaching from cage structures (particularly cadmium, lead, copper and zinc) may be toxic to or accumulate in aquatic biota. Metals are also constituents in fish feed formulations. In a recent study, Chou et al. (2002) determined the level of copper, zinc, iron and manganese in fish feed, fish feed constituents and sediments collected near aquaculture sites in southwest New Brunswick. The concentration of copper in sediment was over two times greater than the concentration found in feed at sites rated as hypoxic or anoxic. Zinc was one to two times greater. These metals showed a positive correlation with the level of organic carbon, and the parameter appeared to be related to hydrographic characteristics at individual sites. Iron concentrations in sediment showed no apparent effect of aquaculture. Manganese concentrations in sediment showed a negative correlation with the level of organic carbon. Chou et al. (2002) concluded that research should be conducted with a goal of developing sediment guidelines for aquaculture activities.

Burridge et al. (1999a) also reported concentrations of metals in sediments collected near aquaculture sites. As stated above, copper concentrations exceeded recommended sediment quality guidelines. Similarly, zinc concentrations in some samples were found to exceed the threshold effects level (CCME 2002). The authors suggested zinc may have contributed to some lethal effects in standard invertebrate bioassays. Cadmium was found to exceed the ocean dumping guidelines of 0.7 $\mu g \cdot g^{-1}$. Lead and mercury were also measured, but levels did not exceed any established thresholds. Mercury has also been measured in fish meal, fish feed and fish oil of various origins (CFIA 2002). The average concentration in fish meal and fish feed ranged from 0.1 to 0.3 $mg \cdot kg^{-1}$. No mercury was detected in fish oil samples. These data, the authors suggest, indicate that it would be unlikely that fish would accumulate mercury levels above the maximum limit prescribed by the Canadian Guidelines for Chemical Contaminants and Toxins in Fish and Fish Products of 0.5 $mg \cdot kg^{-1}$ (CFIA 2002).

Parker and Aubé (2002) reported the results of metal analyses conducted on sediments collected under aquaculture pens. As stated earlier, they reported copper levels exceeded CCME Interim Sediment Quality Guidelines in 80% of the samples and at 9 of 10 sites. Three samples had copper levels that exceeded the CCME probable effects level. The average zinc concentration also exceeded the CCME Interim Sediment Quality Guidelines, although all samples had concentrations less than the CCME probable effects level. Similar to Burridge et al. (1999a), these authors also found several samples where cadmium was higher than the ocean dumping guidelines. Finally, Parker and Aubé (2002) reported nickel, cobalt and iron concentrations were not significantly different from background and did not appear to be of any environmental concern.

DISINFECTANTS

With the recent outbreak of Infectious Salmon Anemia (ISA) in southwest New Brunswick, disinfection of equipment and site infrastructure has become a priority at all aquaculture sites. Torgersen and Hástein (1995) describe the products suggested for use as disinfectants in Europe, as well as the appropriate concentrations and method of application. Disinfection methods range from simply drying (nets and equipment) to treating with chemicals. In Canada, iodophors are routinely used for disinfection of equipment and are unlikely to cause any environmental effects (Zitko 1994). However, the chemical composition of individual formulations has not been determined, and it is known that some may contain compounds such as ethoxylated nonylphenols that may be toxic to aquatic biota (Zitko 1994). Alkylphenols, particularly nonylphenol, have been implicated as endocrine disrupting agents (Madsen et al. 1997; Ashfield et al. 1998).

Chlorine and hypochlorite are used in net cleaning operations and in treating blood water and process plant effluents. Chlorine is very toxic to aquatic biota and, therefore, must be used with caution (Zitko 1994).

Quaternary ammonium compounds have been suggested for use as disinfectants in the New Brunswick aquaculture industry. However, it appears that no one is using these at this time (K. Coombs, New Brunswick Department of Agriculture, Fisheries and Aquaculture, St. George, NB, personal communication).

CONSTRUCTION MATERIALS

Wood construction of cages is no longer common. However, wooden cages with styrofoam floats are still present. Polystyrene 'beads' from these floats are a source of low-molecular weight contaminants (Zitko 1994). In addition to being a source of chemical contamination, these beads can be consumed by fish and birds and are widely deposited on local beaches. Contamination by plastics has become a world-wide problem. Goldberg (1997) states that plastics can alter the nature of benthic communities by altering pore water gas exchange, by ingestion or by providing habitat for opportunistic organisms. The current situation around aquaculture sites is not known, nor is it known whether the aquaculture contribution to local contamination is significant.

Low molecular weight components of plastics may be a source of contaminants both to fish and the environment (Zitko 1994). These may include plasticizers, stabilizers, lubricants, coloring material, antioxidants, UV absorbers, antistatics and flame-retardants. Plasticizers have been shown to bioaccumulate, as have flame-retardants.

ANAESTHETICS

Anaesthetics are used in operational situations when samples are collected for weight measurements, sea lice counts and when handling broodstock. MS-222 (tricaine methanesulfonate) is registered for use by Health Canada (2001) and is used in the local aquaculture industry. No environmental effects are foreseen with its use (Zitko 1994).

OTHER SOURCES

Chemical formulations have been alluded to with respect to pesticide and disinfectant products. Many of the so-called 'inert' ingredients in these formulations may not actually be inert. Without full knowledge of the constituents of each formulation, an accurate assessment of potential risk(s) cannot be made.

The aquaculture industry has historically been concentrated in small coastal bays and inlets. This is particularly true of southwest New Brunswick, where the concentration of aquaculture activity results in considerable small boat traffic in these areas. There are large numbers of outboard motor boats, as well as outboard-driven barges and larger service vessels, used to handle fish and fish feed. It is possible that significant quantities of gas, oil and lubricants may enter the water as a result of the normal operation of these vessels.

RESEARCH NEEDS

The concentrated nature of finfish aquaculture in southwest New Brunswick has made this a prime area for environmental research. The industry in other Canadian jurisdictions, particularly in British Columbia, is spread over wider geographic areas and it appears to have been the subject of very little chemical-related research. Nash (2001) has identified environmental issues related to aquaculture activity in the Pacific Northwest of the United States. Of three issues related to aquaculture he identifies as carrying the most risk to the environment, two are related to chemicals: accumulation of metals in sediments and the potential of therapeutants to affect non-target organisms.

Research Needs:
Research is needed to address issues pertinent to all areas where marine finfish aquaculture is practiced in Canada. Studies are needed to determine the use, presence and effects of chemicals used in aquaculture, particularly in the following areas:

- southwest New Brunswick, where existing knowledge can be expanded and research can be conducted in concert with other environmental effects projects;
- Newfoundland, Nova Scotia and British Columbia, where little, if any, such work has been conducted.

The possibility that an organism will be affected by a chemical, or chemicals, is a function of the toxicity of the chemical(s), probability of exposure and duration of exposure. In a limited number of cases, data regarding the toxicity of the compound(s) are available. This knowledge is, unfortunately, limited to lethality tests conducted over unrealistic time frames. More research is needed to determine sublethal effects and the effects of realistic exposures. Commercially important non-target species have attracted much of the attention regarding effects of chemicals. There are apparently no data regarding the effects of these chemicals on microorganisms and planktonic species in the near-shore environment. These organisms form the foundation of the marine food chain.

Research Needs:

- Ensure pesticides and drugs undergo testing using species native to areas of aquaculture activity. Species should include not only those of commercial importance but also species from several trophic levels.
- Research needs to be conducted to understand the 'normal' physiology of test organisms, so investigators can be confident that work is being carried out on healthy individuals. Research into lethality of pesticides and drugs must include investigations into the effects of repeated exposures, short-term exposures and the effects of season of exposure. These types of tests should be standardized to ensure consistency of application.
- Research needs to be expanded in the area of chronic toxicity of pesticides and drugs.
- Research needs to be expanded into identifying sublethal hazards to non-target organisms of the use of pesticides and drugs in aquaculture.
- New methods of exposing test organisms to pesticides and drugs should be developed to more realistically reflect exposure of non-target organisms in the field. Work may include studies of feeding behavior and food preferences of test organisms and methods of chemical delivery.

All work reported here, and indeed reported in the literature, relies on single species and single compound testing. There is a serious lack of data regarding the cumulative effect(s) of exposure to chemicals of aquaculture origin. Compounds such as POPs, PAHs, PCBs and metals can (and likely do) originate from sources other than aquaculture. The contribution of other sources of contaminants to cumulative impacts is not known.

Research Needs:

- Research is needed to investigate means of assessing cumulative impacts of chemicals.
- Research into lethality of pesticides and drugs should include investigations into the effects of repeated and short-term exposures.
- Effects of interactions of chemicals and other stressors on sensitivity of test organisms should be investigated. These may include stressors such as water temperature, food deprivation, presence or absence of predators, and presence or absence of shelter.
- Laboratory-based mesocosm studies to investigate the effects of aquaculture chemicals should be performed.

The source, concentration and fate of chemicals of aquaculture origin is poorly understood. The persistence of chemicals in sediment and biota, in most cases, is not known. The limited field data reported in this review represent snapshots of local conditions. Interpretation of data is hindered by a lack of information regarding exact sample locations, production figures and treatment history at nearby sites. These data provide the first step in evaluating the persistence of compounds and the risk to non-target organisms. An interesting finding of Burridge et al. (1999a) was the heterogeneity of sediment samples collected by the same diver at the same spot at the same time. This suggests that the under-cage or near-cage environment is not uniform and that sampling protocols for research and/or monitoring should be designed with this in mind. The limited field data available are of little interpretive value because of the difficulties associated with sampling methods and questions regarding how representative the samples are.

Research Needs:

- There are a variety of commercial fish feeds available for marine finfish aquaculture. Research is needed to identify the presence and concentration of contaminants in feed and to determine the fish source.
- Research is needed to determine and develop better field methods for investigating the presence and fate of chemical contaminants in sediments in areas of aquaculture activity. A mechanism must be developed whereby sediment samples can be collected under cages and in areas immediately around cages.
- Models need to be developed and utilized to establish the relationship between normal farm practices (production, feeding and chemical treatment history), local hydrographic conditions and sensitivity of non-target species.
- The presence and prevalence of antibiotic resistant organisms in sediments and indigenous species around aquaculture sites need to be investigated.
- Research is needed to address the remaining research questions about presence and persistence of antibiotics in Canadian sediments.

While there are laboratory-derived data on many of the compounds mentioned in this brief review, there is almost no information regarding effects of chemicals of aquaculture origin in the field situation. Most work to date has been aimed at determining the effects of anti-sea lice chemicals in the laboratory. Limited field trials have been conducted, but these studies have focused on lethality of single treatments. Even these are inadequate in evaluating risks.

Research Needs:

- Field surveys and field experiments need to be conducted. These should include studies of short- and long-term responses of individual species to pesticide and drug applications.
- Long-term studies to establish the natural variability in near-shore marine communities need to be conducted to assess if a chemical-related change has occurred.
- Long-term studies need to be conducted to determine if chemical-related changes in indices, such as biodiversity, occur.

All research into the presence, persistence and effects of chemical contaminants depends on analytical chemistry and requires trained analysts and state of the art equipment. Without this support, many important questions regarding the impacts of aquaculture chemicals will remain unanswered. This is particularly true with regards to risk assessment.

Research Needs:

- Chemical analytical research needs to be conducted in tandem with and in support of work on biological endpoints. This needs to include development of methods for quantifying concentrations of organic and inorganic contaminants in water, sediment and biota. In addition, methods are needed to determine biochemical indicators of health in non-target organisms.

Major questions remain regarding chemical contaminants related to aquaculture and their effects on the marine environment. Information is lacking on environmental trends and underlying ecological mechanisms.

Investigating the potential effects of aquaculture chemicals is essentially an investigation of change. There is, therefore, a need to assess long-term spatial and temporal changes in the vicinity of aquaculture operations. To identify effects or trends, we must understand what is being (or has the potential to be) changed. Changes in biodiversity, for example, cannot be identified in short-term studies. Similarly, investigating effects of compounds on non-target organisms requires an extensive understanding of the organism, its physiology and relationship to its environment. As stated earlier, most of the non-target work addresses effects on important commercial species. It is equally important to develop an understanding of other organisms that may serve as indicators of environmental health. This type of research is long-term in nature.

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