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# Shellfish Aquaculture and Marine Habitat Sensitivity Case Studies

# Études de cas sur la conchyliculture et la sensibilité de l'habitat marin

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## ABSTRACT

The definition of ‘habitat sensitivity’ used in this paper follows the ICES (2002) definition - “Habitat sensitivity can be defined in relation to the degree and duration of damage caused by a specified external factor. Sensitivity may refer to structural fragility of the entire habitat in relation to a physical impact, or to intolerance of individual species comprising the habitat to environmental factors, such as exposure, salinity fluctuations or temperature variation.”

The ‘specified external factor’ in this case is shellfish aquaculture, and the ‘sensitivity’ of marine habitats to this factor is explored via three case studies (eelgrass, large scale intertidal soft bottom and a shallow bay).

Present shellfish aquaculture practices in Canada have the potential to negatively impact sensitive marine habitats. However, these effects can be controlled by managing the intensity of shellfish aquaculture activities on a bay wide scale. Adaptive management informed by ongoing monitoring offers the best route to control the cumulative impacts associated with this industry. The proposed bay wide management scheme offers a positive economic incentive to the industry, as the same cumulative impacts which harm sensitive habitat are those which act as a ‘feed back loop’ to reduce shellfish production (i.e. exceeding the carrying capacity of the local environment to support maximum growth rates of shellfish).



## RÉSUMÉ

La définition de « sensibilité de l’habitat » utilisée dans le présent document est la même que celle du CIEM (2002), à savoir que la sensibilité de l’habitat peut être définie selon l’ampleur et la durée des dommages causés par un facteur extérieur précis. La sensibilité peut désigner la fragilité structurale de tout l’habitat par rapport à un impact physique, ou l’intolérance de certaines espèces qui composent l’habitat à l’égard de facteurs environnementaux tels que l’exposition, les fluctuations de la salinité ou la variation de la température.

Dans le présent cas, la conchyliculture est le « facteur extérieur précis » et la « sensibilité » des habitats marins à ce facteur est examinée au moyen de trois études de cas (zostère marine, fond mou intertidal à grande échelle et baie peu profonde).

Les pratiques actuelles en matière de conchyliculture au Canada peuvent avoir une incidence négative sur les habitats marins vulnérables. Toutefois, ces effets peuvent être limités par la gestion de l’intensité des activités de conchyliculture à l’échelle d’une baie. La gestion adaptative alimentée par une surveillance continue constitue la meilleure méthode pour régir les effets cumulatifs liés à cette industrie. Le plan proposé de gestion à l’échelle des baies offre un stimulant économique pour l’industrie, puisque les mêmes effets cumulatifs qui nuisent à l’habitat vulnérable agissent « en boucle » pour réduire la production conchylicole (c.-à-d. en excédant la capacité biotique du milieu local à soutenir des taux maximaux de croissance des mollusques).

# INTRODUCTION

## Definition of 'Sensitive Habitat'

Marine habitat can be defined as a set of physical, chemical and biological conditions which are conducive to the survival of a population of organisms. The organisms use that particular marine space for all or part of their life history for the purposes of feeding, migration, refuge, reproduction, etc[1].

A consensus was reached during the national finfish aquaculture peer review meeting (Institute of Ocean Sciences, Sidney, BC – February 22 to 25, 2005) to use the ICES definition for habitat sensitivity:

> "Habitat sensitivity can be defined in relation to the degree and duration of damage caused by a specified external factor. Sensitivity may refer to structural fragility of the entire habitat in relation to a physical impact, or to intolerance of individual species comprising the habitat to environmental factors, such as exposure, salinity fluctuations or temperature variation." (ICES 2002)

The objective of this working paper is to examine the 'sensitivity' of selected marine habitats (eelgrass, large scale intertidal soft bottom and a shallow bay) against the effects of shellfish aquaculture as an 'external factor'. The case study approach is used.

# CASE STUDY #1 – EFFECTS OF OYSTER AQUACULTURE ON EELGRASS (Herb Vandermeulen)

## Introduction

In order to provide a management context for this case study, four interrelated facts need to be discussed:

1. *Zostera marina* L. (eelgrass) is a sensitive habitat;
2. both eelgrass and oyster reefs are important habitat;
3. seagrasses like eelgrass offer a settlement site and haven for bivalves;
4. bivalve aquaculture (including oyster culture) has the potential to harm eelgrass

[1] The *Fisheries Act* is quite specific in its definition, S34.(1) - "fish habitat" means spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes

## Eelgrass as sensitive habitat

In 2005, DFO Science used a formal, peer reviewed process to state that eelgrass is a sensitive habitat (Vandermeulen 2005). *Zostera* is a vascular marine macrophyte found rooted in sandy or muddy substrates on all three Canadian coastlines (den Hartog 1970). The plants can form extensive subtidal, perennial beds widely recognized as important nearshore habitat for juvenile (and adult) invertebrates and fish (Short and Wyllie-Echeverria 1996; Chambers et al. 1999). The beds provide cover from predation, reduce local current regimes (allowing for settlement of organisms) and increase secondary productivity by adding to local habitat complexity and surface area (Chambers et al. 1999; Boström et al. 2002;  Duarte 2002; Laurel et al. 2003).

## Oyster reefs as habitat

There is no doubt that a collection of oysters on a natural bottom may reduce erosion (i.e. protect existing benthic habitat), help increase water clarity via filtration of the water column, provide a vertically structured habitat and offer valuable shelter for marine organisms (Dumbauld et al. 1993; Meyer et al. 1997; Coen et al. 1999; Coen and Luckenbach 2000; Breitburg et al. 2000; Cressman et al. 2003; Peterson et al. 2003; Escapa et al. 2004 ; Nelson et al. 2004; Luckenbach et al. 2005). Logically, a case can be made that artificially structured oyster populations (e.g. oyster aquaculture) also provides useful habitat (Breitburg et al. 2000; Meyer and Townsend 2000 ; Rheault 2001).

However, the value of oyster reefs as habitat should not be used as an excuse to artificially promote their presence over wide areas of the bottom (i.e. intensive oyster aquaculture). A spatially varied mix of soft bottom, marine macrophyte, salt marsh, oyster reef and clam bed benthic habitat types offers greater ecosystem function than a monoculture of oysters (Breitburg et al. 2000)[2].

Castel et al. (1989) compared ‘oyster parks’, rack and direct bottom culture oyster aquaculture areas, to *Z. marina* beds in Arcachon Bay, France (an area of intensive oyster aquaculture)[3]. The eelgrass beds held greater abundances of both meio- and macrofauna. Macrofaunal densities were particularly depressed in the oyster park area, in part due to low oxygen levels induced by the high concentrations of oysters.

Dealteris et al. (2004) examined the habitat value (abundance and diversity of associated marine organisms) of oyster rack aquaculture gear (a more three dimensional structure than oyster reefs) and determined that its habitat value was at least equal to eelgrass. In a study of decapod crustacean use of estuarine habitats, Glancy et al. (2003) found that decapod density was similar between

[2] The oyster species found in temperate waters all have a similar ecological role and habit. In Canada, natural oyster reefs tend to be found up drainage channels in low salt marsh areas. We have *Ostrea lurida* on the west coast, and *Crassostrea virginica* in the east, along with cultured species like *Crassostrea gigas* and *Ostrea edulis.*

[3] The different types of oyster aquaculture are described in the ‘Oyster Aquaculture’ section below.

seagrass and natural oyster reef habitats while the composition of the decapod species was different. Juvenile hard clams (*Mercenaria mercenaria* L.) survive better in oyster shell hash environments compared to seagrass beds (Peterson et al. 1995).

Heck et al. (2003) reviewed over 200 papers on seagrasses as nursery areas. They concluded that seagrasses are important nursery areas, especially in the northern hemisphere. Other structured habitats like oyster reefs, cobble reefs and macroalgal beds were found to have similar value to seagrasses as nursery areas.

In contrast to the studies mentioned above, the review by Minello et al. (2003) indicated that seagrass was superior to oyster reef as habitat. Both fish and decapod crustaceans had higher density in seagrass.

Hosack et al. (2005) conclude that habitat use coincides with the size of an organism and its mobility. Smaller benthic organisms appear to be affected more by the presence of structured habitat (e.g. seagrass bed or oyster reef) than larger highly mobile organisms like crabs or fish. The relative ecosystem value of oyster versus seagrass habitat may then be a matter of scale.

## Seagrass affects on bivalves

Seagrass attracts bivalve settlement (Reusch 1998; Bologna et al. 2005). The presence of seagrass increases bivalve settlement at different spatial scales, from epiphyte microstructure on the blades, to the configuration of the beds themselves (the edges of the beds generally allowing for greater settlement, Bologna and Heck 2000). Hydrodynamics also appear to play a role (Eckman 1987 ; Grizzle et al. 1996).

Seagrasses also influence bivalve predation rates and growth rates. Bologna and Heck (1999) found *Argopecten irradians* Lamarck (bay scallop) more abundant associated with turtle grass (*Thalassia testudinum*) than unvegetated sediments, and particularly abundant at the edges of the turtle grass beds. Predation and growth rates of the bay scallop were lower within the turtle grass beds, and higher at the edges of the beds. The protection the seagrass beds provided came at the cost of lowered growth.

Irlandi et al. (1995) obtained similar results for bay scallop in mixed *Halodule wrightii* Ascherson and *Z. marina* seagrass beds, with greater predation rates in more patchy beds. The hard clam (*Mercenaria mercenaria* L.) also enjoys predation protection within these seagrass meadows, with greater protection in larger beds (Irlandi 1997). Heck et al. (2002) found that long term growth of *Mercenaria* was not reduced in seagrass beds compared to unvegetated areas.

## Oyster aquaculture

Oyster aquaculture can be broken down into phases of seed collection, grow out and harvest (Kaiser et al. 1998). Each phase can have environmental impacts, depending upon the intensity of the activity. Seed collection is usually benign;

often involving passive spat collectors. The grow out phase is accomplished by placing young oysters in different configurations and densities (Simenstad and Fresh 1995; Kaiser et al. 1998):

1. direct bottom culture (ground culture) – most common culture method, young placed directly on bottom, usually at low density (Simenstad and Fresh (1995) cite “light shell” as 10 – 50% cover of oyster and “heavy shell” as >50% cover)
2. stake culture – stake or other support hammered into the bottom, oysters suspended on the stake (e.g. in bags) several feet off the bottom, or in bags on a line between stakes (longline culture), usually intertidal – moderate / high density
3. rack culture – oysters placed on racks (or ‘tables’) held on or above the bottom by posts, usually intertidal – moderate / high density
4. float culture (suspended culture) – used in deeper water, oysters suspended vertically on lines supported by floats (Crawford et al. 2003)

Harvest phase methods:

1. direct bottom culture - hand collection (SCUBA is sometimes used), tongs, rakes, or dredging (Lenihan and Peterson 2004)
2. stake or rack culture – usually hand collection at low tide, often by driving vehicles onto the tidal flat

The impacts on sensitive habitat associated with oyster aquaculture depend upon density of oysters on the site, extent of handling / sorting during grow out and the harvest methods used (Kaiser et al. 1998). For example, low density direct bottom culture coupled with SCUBA based sorting and harvest is expected to have far less impact than high density direct bottom culture involving large scale sorting and transfer during grow out and harvesting by dredge.

Lenihan and Peterson (2004) demonstrated that a switch from tong / dredge oyster harvest methods to SCUBA acted to protect oyster reef integrity (measured as changes in reef height and diameter compared to controls). This implies that harvest method can also modify the habitat value of the oyster reef itself.

## *Chemical Effects*

### Nutrient loading

Oyster aquaculture does not spontaneously create a new nutrient source in an aquatic system. However, the oysters act to sequester particulate nutrients (i.e. their planktonic food) and subsequently excrete nutrient rich fecal pellets that rapidly sink to the bottom, and dissolved nutrients (primarily nitrogen in the form of ammonia). The net effect is nutrient enrichment of the surrounding sediment (and related changes in geochemistry, including negative redox potential and reducing conditions), and the release of nitrogen back into the water column (e.g. Barranguet and Alliot 1995).

Crawford et al. (2003) examined three oyster suspended culture facilities in Tasmania. The facilities were located in approximately 10m of water, growing mainly oysters with some mussels. Redox, organic carbon and sulphide measurements indicated some alteration of sediment geochemistry at two of the sites consistent with organic enrichment – although the authors did not consider this excessive. Black sediment and bacterial mats (*Beggiatoa* sp.) indicative of anoxic conditions were noted at one of those sites. The site with the 'best' sediment quality also had dense beds of the seagrasses *Heterozostera tasmanica* and *Halophila australis.* The authors did not note the presence of seagrass at the two sites with poorer sediment quality.

Villarreal (1995) describes eutrophication and organic enrichment of sediment associated with oyster culture (type not specified) in Baja California. The effect seemed quite localized, and nearby eelgrass meadows were not affected. In another study in Baja, Ward et al. (2003) were also not able to correlate oyster cultivation (rack style) with eelgrass loss.

Mojica and Nelson (1993) document near field reductions in the seagrasses *Halodule wrightii* Ascherson and *Syringodium filiforme* Kützing by a hard clam (*Mercenaria mercenaria* L.) grow out facility in Florida. USA. *Mercenaria* are somewhat similar in size to oysters, and in this particular instance direct bottom grow out procedures were used (approximately $1m^2$ mesh bags full of clams placed directly on the bottom). Sediment geochemistry was noted as more reducing at the clam grow out site compared to controls, along with smaller grain size and more volatile solids. The shallow waters at the study site were well mixed, and the authors did not find any consistent differences in oxygen, nutrients or phytoplankton. No consistent significant differences in benthic organisms were seen. The reason for the reduced seagrass abundance at the farm site is unexplained in the paper, but may be related to the recorded alterations in sediment geochemistry (seagrasses have difficulty growing in anoxic, reduced sediments - Vandermeulen 2005).

## Low oxygen levels

Plus et al. (2003) report on eutrophication generated by a mixed mussel / oyster aquaculture site in the Thau Lagoon (South France) which caused bottom water anoxia for four days, destroying local eelgrass meadows. The seed bank allowed the beds to recover.

The anoxia came from the aquaculture activity, and "The triggering factor was the degradation of green algae and probably organic matter coming from aquaculture…" De Casabianca et al. (1997) determined that shellfish farming was the major source of eutrophication in the lagoon, providing nitrogen loading rates about 15 times the terrestrial inputs to the system. Loading came in the form of biodeposition and ammonia excretion from the mixed oyster / mussel biomass grown on site.

*Zostera* (*Z. marina* and *Z. noltii*) were likely the original plant community in Thau Lagoon. With increasing eutrophication from shellfish aquaculture, *Zostera* was replaced by "opportunistic and nitrophilous species" *Ulva* and *Gracilaria.* In effect, the *Zostera* was "pushed" into less nutrient laden portions of the lagoon by aquaculture activities (De Casabianca et al. 1997b; De Casabianca et al. 2003).

Barranguet and Alliot (1995) examined benthic fluxes of oxygen and ammonia in the extensive oyster farming area of the Thau Lagoon and discovered that ammonium fluxes from the sediment and the abundance of microphytobenthos were higher under the oyster cultivation units (metal frames with lines holding the oysters). The observations are consistent with a local eutrophication of surficial sediments caused by the oyster culture. They also noted a general tendency towards anoxia in the oyster cultivation area. Deslous-Paoli et al. (1998) consider the resulting water column anoxia to be an 'ecosystem dysfunction' which detracts from the positive value of the oyster biomass on the Thau Lagoon ecosystem (uptake of particulate material, storage of nutrients in animal tissue, development of the benthos, etc.).

Low sediment oxygen levels were also associated with high density oyster cultivation (rack and ground culture) in Arcachon Bay, France (Castel et al. 1989). The accumulation of biodeposits and subsequent alteration of sediment geochemistry towards hypoxic conditions was cited as the cause.

### Pesticide application

The pesticide carbaryl is applied to control burrowing shrimp (thalassinids) in Willapa Bay, Washington, a major oyster aquaculture area[4]. The shrimp are strong bioturbators and negatively impact the oysters (Dumbauld and Wyllie-Echeverria 2003). Interestingly, the reduction in bioturbation upon pesticide application supports the growth of *Z. marina* and *Z. japonica.*

## *Biological Effects*

Although oyster aquaculture has been associated with the introduction of alien species (e.g. Britton-Simmons 2004), the author could not find an example of an invasive linked to oyster aquaculture that would negatively impact eelgrass. *Crassostrea gigas* culture on the west coast introduced both *Sargassum muticum* (a brown alga) and *Zostera japonica*, but these plants do not widely overlap eelgrass distribution. There is a similar lack of evidence for oyster aquaculture impacting eelgrass by altering herbivore abundance, introducing pathogens, or affecting fouling.

[4] This practice may not continue for long. The Willapa Bay / Grays Harbour Oyster Growers will discontinue the use of carbaryl by the year 2012 (Brett Dumbauld, pers. comm.)

### *Physical Effects*

The husbandry and harvest methods used for oysters can have a variety of physical affects, including alterations in light levels, changes in local currents and the incidence of scour, and sedimentation. For example, bivalves can filter out substantial amounts of phytoplankton on a bay wide scale (Cloern 1982; Officer et al. 1982) and there is some evidence that the filtration efforts of oysters can improve water clarity to the point that seagrasses benefit (Newell and Koch 2004).

Simenstad and Fresh (1995) describe negative effects on *Zostera marina* in the presence of oyster culture related to:

- removal of the plants to alter water flow and ease harvest by dredging
- dredging, harrowing and levelling of oyster plots related to substrate preparation for grow out, and harvest

The longer an oyster plot was used, the more extensive and persistent the loss of eelgrass. Both bottom culture and stake and rack culture of oysters was associated with substantial eelgrass loss (Simenstad and Fresh 1995).

Brett Dumbauld (US Department of Agriculture, Oregon) recently completed a multiyear study to determine the impacts of oyster culture on eelgrass. Although the results were variable they were "…able to show a consistent trend in the effects of harvest practices with reduced eelgrass density in all areas where oysters were cultivated and approximately two thirds lower density observed in areas where a harvest dredge implement had been used versus that found in nearby eelgrass meadows. Beds where oysters were picked by hand and where long-line culture was used had intermediate densities and cover." (Dumbauld 2005 unpublished report). Eelgrass could recover within 1-2 years if left alone, moreover:

- long-line culture experiments[5] indicated that increased spacing between lines could reduce impacts (most significant impacts occurred at spacings < 2.5 feet)
- oyster bottom culture had an apparent effect on eelgrass density, but only at high planting density (300 seed bags/acre) and only on muddy substrate. No effects were observed on eelgrass growth.

In an earlier Oregon based study, Everett et al. (1995) examined stake and rack culture of oysters. Both culture methods negatively impacted *Zostera marina*, reducing cover to less than 25% of reference plots after one year. Plants were absent from the rack treatment after less than two years. Stake culture was associated with increased sedimentation and physical disturbance of the plants during oyster placement and harvest; while rack culture lead to increased erosion and shading of plants.

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[5] Rumrill and Poulton (2004) provide some more details, including the fact that spacing between lines may have to reach 10 feet before cover and density of eelgrass at the aquaculture site are within the range of variability seen at reference sites.

Wisehart et al. (2004) report on oyster cultivation methods and eelgrass abundance. Longline culture supported eelgrass biomass and growth similar to reference areas. Eelgrass growth and biomass in handpicked ground culture and dredged culture areas were reduced, and to similar levels. They suggest that site-specific conditions may be as influential as culture technique in determining eelgrass growth.

De Grave et al. (1998) studied an intensive stake / rack style (trestle) intertidal oyster aquaculture site in Ireland and concluded that organic enrichment of the area did not occur, mainly due to water exchange. They did, however, note impacts on the macrofauna in the access lanes between trestles due to heavy vehicle traffic.

The impacts of trestle oyster aquaculture were also studied in England. In this case, organic enrichment did occur under the trestles and the macrofauna changed (Nugues et al. 1996). The trestles themselves may have slowed currents and increased sedimentation rates.

### Some examples of effects from clam harvesting

Although clams are infauna and oysters are usually on the surface of sediments, there are some examples of effects from clam harvesting that are relevant to our discussion. Traditional clam harvesting in southern Portugal reduces shoot density in *Zostera noltii* and increases the reproductive effort (seeds, flowering shoots), a sign of stress. The hand blade clam harvest breaks and removes the shoots and rhizomes of the plants (Alexandre et al. 2005). Recovery can be rapid, depending upon the extent of damage to individual plants (Cabaco et al. 2005).

In comparison, an experimental attempt to determine impacts of recreational clam harvest in an eelgrass bed in Oregon was equivocal (Boese 2002). It appears that the intensity of clam harvest activity (and concomitant eelgrass disturbance) can vary widely.

Badino et al. (2004) report on harvest methods for the Manila clam (*Tapes philippinarum* Adams and Reeve 1850) in the Venice Lagoon. Use of a propeller wash / mechanical dredge method for harvesting the clams created a reduction in benthic habitat quality indices, including sediment compaction, decrease in the abundance of macrobenthic organisms and depletion of the oxidized sediment layer. Using a modelling approach, Pranovi et al. (2003) predicted a 33% increase in artisanal fishery catches (a traditional fyke net fishery) if clam dredging was eliminated from the lagoon. While this is an extreme example (dredging into the sediment for an infaunal organism), it does point to potential problems with this method if applied to sediment surface bivalves like oysters.

Dredging for a sediment surface bivalve (*Mytilus edulis* L.) in Denmark appeared to create minimal impacts (reduced density of polychaetes and predators attracted to the disturbed area). The dredge furrows were quite shallow (2-5cm) and the infrequent commercial dredging of individual mussel beds needs to be considered against normal background disturbance to the beds (Dolmer et al. 2001).

## Effects of oyster aquaculture on seagrasses - summary

It is difficult to provide a quantitative summary of the effects of oyster aquaculture on seagrass. The studies noted above rarely specify actual oyster density, or other details of the culture system. Effects appear to be cumulative and from multiple factors. Organic enrichment of sediments is linked to alterations in geochemistry, but also against a spatial mosaic due to site specific erosional / depositional patches created by the arrangement of the aquaculture infrastructure. In general, seagrass appears to suffer when oyster aquaculture is present, but site specific differences in current regime, gear use and harvest method may go a long way to ameliorating these effects.

From a scientific perspective, it is tempting to sort out which factors (direct and indirect effects of nutrient loading, low oxygen levels, light levels, currents and scour, sedimentation, etc.) are responsible for specific aspects of seagrass decline in the face of aquaculture (as in Vandermeulen 2005). However, this approach is not feasible with the information at hand and may not be particularly useful to managers of the oyster industry at this time.

Adaptive management, or experimentally based management, may be the best path in this instance (Chambers et al. 1999). This is explored in the conclusions section below.

# Conclusions

Given the evidence at hand, the Pregnall's conclusion should be used as guidance - oyster culture should not be allowed in areas of eelgrass meadows (Pregnall 1993, cited by Simenstad and Fresh 1995). If the two are to coincide (which is likely given the salinity and temperature preferences of oysters and eelgrass), a context must be given for the decision making process regarding oyster culture:

1. eelgrass is a sensitive habitat – Sensitive habitat should be preserved. The author suspects that natural oyster reefs in Canada are also sensitive habitats, but is unaware of an 'official' statement from DFO Science to this effect.
2. oyster reefs are significant habitat – Seagrass beds and oyster reefs both appear to haven important habitat value, although they may attract and support different organisms.
3. seagrasses offer a settlement site and haven to bivalves – The presence of eelgrass will be beneficial to bivalves.

So, a mix of eelgrass beds and natural oyster reef would offer a high quality marine habitat where the presence of the eelgrass would be beneficial to the oysters. The addition of an oyster aquaculture operation to such a site would need to preserve the existing habitat (due to its sensitive and important status as habitat). The preservation of the existing habitat could be achieved by developing oyster aquaculture on site as follows:

1. density of added oyster biomass should be low –
    a. direct bottom culture: <10 to 50% cover of oyster (Simenstad and Fresh 1995), <300 seed bags/acre (Dumbauld 2005 unpublished report)
    b. long-line culture: >10 feet spacing between lines (Rumrill and Poulton 2004)
    c. stake, rack and float culture: unknown at present, but an industry estimate of the meaning of ‘low density’ for each of these culture methods could be used
2. configuration of aquaculture gear (stakes, racks, etc.) should be such that excessive erosional or depositional ‘patches’ are not created
3. harvest method should avoid bottom disturbance as much as possible –
    a. driving heavy machinery onto the intertidal should be prohibited
    b. dredge harvest should be prohibited, and use of tongs avoided. SCUBA harvest is preferred

Once oyster aquaculture has been established on site, monitoring of sediment, eelgrass and native oyster health should occur for at least two years prior to the expansion of aquaculture operations. Expansion should be iterative, based upon proof of no harm with each iteration.

# CASE STUDY #2 – INTERTIDAL SHELLFISH AQUACULTURE IN BAYNES SOUND (Glen Jamieson)

## Intertidal zone as sensitive habitat, relative to other fish habitat

In this case study to explore the sensitivities of habitat to shellfish aquaculture, I would first like to define the terms “habitat” and “sensitivity” in the context that I will discuss them. Firstly, I consider habitat to be more than just physical substrate, and include in this term the biological communities and species that would normally occupy the physical location in the absence of aquaculture. In this sense, a better term would be ‘ecosystem’, so I will be considering the impacts of aquaculture on local and regional ecosystems. I also define the ecosystem that might be being impacted by shellfish aquaculture as more than just fish habitat, and consider the habitat of other species, e.g., other bivalves and birds in particular, and indeed, do not even discuss fish in this case study. With respect to sensitivity, I will consider it with respect to birds in the context of whether the aquaculture impact has major regional or global impacts on the spatial distribution or dynamics of regionally-recognised important species. Any effort to achieve a monoculture of a desired commercial species obviously has a local impact, but whether this is acceptable is determined by both the scale of the impact, its nature and the reversibility or “duration” of the impact. If, for example, it impacts a Species at Risk Act (SARA)-listed species, then even a small scale impact may be deemed unacceptable, but in most cases, some acceptable balance between scale of commercial operation and scale of impact on other species will be determined. Here, I look at whether or not an acceptable balance on science grounds seems to have been achieved between shellfish culture and ecosystem impact in Baynes Sound, Strait of Georgia, British Columbia (BC).

### *Background*

There have been few scientific studies of the environmental impact of shellfish aquaculture in the Pacific north-east. The majority of aquaculture studies have focussed on the effects of net pen finfish farms, and of the few studies on shellfish aquaculture, most have revolved around off-bottom culture techniques (WGEIM 2000). Because of the dependence of shellfish aquaculture production on high water quality, it has been assumed as having few environmental impacts. The most notable management issues to date have centred around land use conflicts with adjacent upland owners, recreational harvesters, wild harvesters, other recreational activities, and navigation (deFur and Rader 1995).

Ecosystem concerns have been published regarding intertidal bivalve bottom culture practices (e.g. Simenstad and Fresh 1995), and the scale of existing and planned expansion of this industry in BC has raised concerns among both DFO and BC Ministry of Agriculture and Lands [formerly the BC Ministry of Water, Land and Air Protection (WLAP)] resource managers, particularly in Baynes Sound. Operational activities in Baynes Sound including the delineation of lease areas through the use of Vexar® netting berms, to help retain substrate and/or oysters, the use of predator exclusion nets on many beach surfaces, site-specific modification of substrate and sedimentation characteristics, the tilling of beach surfaces for the thinning and harvest of stock (this may be more frequent on a farmed site than with a wild harvest area rotation strategy), and the channelisation of estuaries, any of which can have either direct or indirect environmental impacts. It should also be noted that most of the intertidal tenures in Baynes Sound farm a mixture of species including both bottom culture of oysters and clams. In a 2001 aerial lease survey, 78 of the total of 106 intertidal tenures in Baynes Sound had clam netting at the time of the survey. 2001 Baynes Sound shellfish tenures produced 3360 t of product: 850 t of clams and 2510 t of oysters (Baynes Sound Coastal Plan, 2002).

Elsewhere, culture practices have impacted the biodiversity and productivity of the intertidal by altering the compositions of benthic intertidal communities, and have excluded some species from foraging areas; reduced the sizes of some finfish spawning, nursery and rearing habitats; and altered the natural coastal hydrography (Simenstad and Fresh 1995). Given a lack of relevant data for Baynes Sound, there was, and still are, public and resource manager concerns that bivalve culture impacts could be affecting the growth and survival of transient fish and wildlife, such as juvenile chinook, coho, chum, pink and steelhead salmon; herring; and, migratory waterfowl and local shorebirds.

The recent bivalve culture management history in BC is that in November 1998, the British Columbia Assets and Land Corporation (MAL) and Ministry of Agriculture, Food and Fisheries (MAFF) introduced the Shellfish Development Initiative, with the goal of increasing the diversification and stability of coastal and First Nations' economies through the expansion of the shellfish aquaculture industry. The 10-year plan allowed a doubling of the farmed area for BC as a whole by roughly 10% per year, but the process is application driven. Operationally, a culture area increase of 99 hectares (16.7%) has been approved in Baynes Sound since 2001 (Nov 1983, 92 tenures = 541.6 ha; Nov 2001, 111 tenures = 594 ha; and Nov 2005, 124 tenures = 693 ha (B. Carswell, Aquaculture Development Branch, BC Min. Agriculture and Lands, Victoria, BC, pers. comm.). All recent rezoning applications for new deepwater sites were rejected by Island Trust, which rejected rezoning for all new sites except for 55 ha for First Nations. Environmental assessments, sent to DFO by Transport Canada, the lead Responsible Authority because suspended culture needs a Navigable Waters permit, of new or expanded shellfish aquaculture proposals have been, or are being, reviewed by the Habitat Management Division (HMD) of DFO. In the absence of previous scientific study of this issue, HMD requested assistance in conducting these reviews. In the last eight years, five new deepwater sites (requiring a Navigable Waters Protection Act (NWPA) permit and therefore a Canadian Environmental Assessment Act (CEAA) screening) have been issued in Baynes Sound. Although HMD had concerns that the projects could add to the cumulative effects in the Sound, it concluded that they would not likely cause significant environmental effects based on the adaptive management approach outlined in the Aquaculture Site Referral Process: Interim Operational – Policy Guidelines (DFO, February 2001). The Interim Policy states that "In such cases, based on the information available at the time of the screening, if it cannot be concluded that the project will likely cause cumulative effects, such effects will not be considered for purposes of preventing a project from proceeding pursuant to s. 20 of CEAA". The Canadian Wildlife Service (CWS) and WLAP expressed concerns about the proposed shellfish lease expansion and the potential impacts on species they are mandated to manage. Given the relatively large number of existing aquaculture leases already present, the cumulative effects of proposed new leases for Baynes Sound needed to be considered.

In response, Science Branch produced a report (Jamieson et al. 2001) that reviewed the scientific literature on environmental impacts of intertidal bottom culture on coastal ecosystem processes, specifically relating to fish and fish habitat; described the current practices of intertidal bottom culture operations and their potential impacts in Baynes Sound; assessed the need for monitoring and/or a cumulative effects study related to the planned increase in leased area in the intertidal zone of Baynes Sound; identified gaps in the understanding of ecosystem impacts of extensive, intensive intertidal bottom bivalve aquaculture; and recommended future research needs.

Jamieson et al. (2001) noted that studies on the habitat impacts of shellfish culture are relatively few and those available were limited in scope and rigour. The literature is still fragmented in its relevance (e.g. Rumrill and Poulton (2003) on the impacts of oyster culture in Humboldt Bay, CA), and while some studies are currently underway, much available information has not yet been scientifically reviewed and published. Views expressed thus remain more hypothesis-generating than definitive, which warrants a need for rigorous testing and evaluation. They brought together available knowledge of the spatial distribution and abundance of the macro-fauna and flora, and pointed out that existing data was unable to address the nature of possible impacts of intertidal aquaculture on the broader ecosystem, i.e., beyond the borders of the immediate area under aquaculture tenure. They also described many of the impacts that were occurring, and the concerns about what possible effects might be on productivity, community structure, juvenile salmonid habitat, herring spawn habitat, and birds, recognizing that shellfish aquaculture has the potential to negatively impact intertidal ecosystems in a variety of ways. In Baynes Sound, eel grass beds may not be being too impacted, as in 1995, only 4.8% of the eelgrass bed area was contained within any shellfish tenures and overlap of eelgrass beds and clam predators netting was negligible (Durance1996). Jamieson et al. (2001) also considered cumulative effects issues, and stated:

> "In order to determine whether an impact has taken place, before-farming baseline conditions must be known and compared with post-farming conditions, or the latter are compared to appropriate reference sites. Because of the dynamic nature of many intertidal areas, it is often unclear as to whether the extent of disturbance from any activity exceeds that to

which biological communities might be normally experiencing and to which they are adapted. There must also be an understanding of the disturbances taking place. As pointed out by others (e.g. Sousa 1984, Simenstad and Fresh 1995), disturbances are not uni-dimensional. Scales include areal extent, intensity (magnitude), local and regional frequency, predictability, and rotation period. It is important to distinguish between the anthropogenic disturbances previously described [in Jamieson et al. (2001)], and natural disturbance regimes such as climatic cycles, storm events, and possible impacts of exotic species or outbreaks of disease (Simenstad and Fresh, 1995). There must also be an understanding of the threshold levels, responses, and recovery times of the environmental impact under consideration. The complex interaction of multiple species, habitats and disturbances on the indicator species needs to be understood. These may include not only direct impacts on the intertidal zone, but also indirect, secondary and synergistic effects. For example, anthropogenic factors from land use activities on the surrounding landscape (e.g., increasing urbanization of upland areas around Baynes Sound) that may contribute secondary impacts to the cumulative effects of intertidal aquaculture in Baynes Sound include:

- changes in water quality from terrestrial land use, increased nutrient loads from fertilisers and pesticide contamination from agriculture, increases in turbidity related to increases in erosion from forestry impacts;
- increased faecal coliform levels from both agriculture and residential septic systems, and also birds and marine mammals, especially sea lions which is a contamination concern in localised areas of the sound, e.g. haulouts in Fanny Bay and Mud Bay in spring during herring season; and
- changes in freshwater input through altered hydrologic regimes from dams and reservoirs, changes in runoff due to altered land cover by forestry and urban development (stormwater runoff) and so on."

The general recommendations from Jamieson et al. (2001) were:

1. to establish a collaborative, inter-agency approach to identify both the nature of existing and potential future impacts and, where necessary, to determine how they can be minimised;
2. to establish an effective network of protected areas in Baynes Sound that includes sensitive habitats and key bird habitat and excludes shellfish culture to serve as both reference sites for future research studies and as "insurance" areas to help ensure that given existing limited knowledge, the natural ecosystem would hopefully be sustained [if it was later found that significant impacts from intertidal aquaculture were occurring];

3. that ocean management in Baynes Sound should be considering intertidal aquaculture both as an economic asset and as an ecological disturbance that may be negatively influencing important ecosystem processes (i.e., productivities of other important species), and that discussion of how balance might be achieved should be occurring;
4. that with increasing bivalve culture (intertidal and suspended) in Baynes Sound, the overall carrying capacity of the system with respect to phytoplankton production and its removal by filter-feeders needs investigation to ensure that the area's carrying capacity is not being exceeded, with resultant decreases in the growth or reproductive rates of farmed or wild filter feeders.

To date, the above first three recommendations have not been acted on by Fisheries and Oceans Canada or by any provincial ministry. The BC Ministry of Agriculture, Food and Fisheries (MAFF) contracted an analysis of Baynes Sound carrying capacity (Hay & Co. Consultants Inc. 2003). Through modelling, the cultured bivalve clearance time was estimated to be 34 days or longer, and since this is much greater than the estimated primary production time of about two days, commercial bivalve populations were suggested to be well below the filter-feeding carrying capacity of Baynes Sound (Hay & Consultants Inc. 2003). However, while this analysis did attempt to consider phytoplankton grazing by zooplankton, it did not consider the filter-feeding biomass of species or individuals outside of farmed areas, which can be expected to greatly exceed the cultured biomass. Thus, utilisation of the carrying capacity of the system arising from culture practices (e.g., abundance enhancement of both commercial and non-commercial filter feeders) has still to be assessed, and specifically, how much does shellfish aquaculture actually increase the extent of filter feeding in Baynes Sound above that expected to occur in the absence of shellfish culture. To phrase this another way, is there a decline in bivalve growth rates at non-culture sites, and if so, can this be correlated with increased local bivalve culture?

There is still concern that intertidal aquaculture may be significantly impacting the broader ecosystem, i.e., beyond the borders of the immediate area under aquaculture tenure. The sensitivity of this larger area and the species that inhabit it to aquaculture activities are the main issues that I attempt to address in this case study. I must first state that any habitat disruption has a recognised local impact, usually acceptable if it addresses our human needs, recognizing that as part of the ecosystem ourselves, meeting our reasonable needs as humans is a totally justified requirement. However, if the impact area is relatively large or extends much beyond the immediate area, then the impact needs to be accessed more broadly to determine if it is having negative environmental consequences to both humans and the broader ecosystem that may be exceeding the value derived from the local impact.

The main recommendations from Jamieson et al. (2001) tried to address this need, as this could not be assessed with the data that was available then. It was in this context that some of the co-authors in Jamieson et al. (2001) initiated follow-up studies with NSERC funding that have investigated bivalves (Leah Bendell-Young (Simon Fraser University (SFU), Burnaby, BC, pers. comm.) and scoters (Dan Esler, Simon Fraser University, Burnaby, BC., pers. comm.):

1. bivalves:
   a) does aquaculture as practiced enhance the abundance of farmed clams,
   b) is there evidence of competitive exclusion among clams within predator refuges created by clam netting, and
   c) are native bivalves affected by aquaculture of a single non-native clam?
2. scoters:
   a) is there modification because of aquaculture of food availability for scoters, through either increases or decreases in food abundance or accessibility,
   b) is there scoter disturbance associated with aquaculture industrial activity, and
   c) does intertidal aquaculture create changes to water quality that affect scoters?

Finally, Bendell-Young (in press) is now publishing her initial work from 2000, which was referred to in Jamieson et al. (2001). To assess the impact of intensive clam shellfish farming on intertidal diversity in Baynes Sound, several indices of ecosystem structure and select geochemical characteristics were contrasted among three geographically similar intertidal regions that represented a gradient of shellfish farming activities; 1) no active aquaculture, 2) actively farmed for three years and 3) actively farmed for five years. The intertidal regions which had been used for farming for three and five years had reduced species richness, a different bivalve composition, abundance and distribution and a foreshore community dominated by bivalves, as compared to the intertidal region where no active farming occurred. Beaches that were actively farmed also had increased accumulations of organic matter and silt. It was noted that simplification of the intertidal benthic community coupled with accumulations of organic matter and increased siltation could compromise the ecology the foreshore region used for intense shellfish harvesting. She suggested studies were needed to determine the scale for which intensive use of the foreshore for shellfish purposes alone was feasible without undue harm to the environment.

Results of these recent new studies are in the process of being published, but salient points are presented below. A caveat to these new studies is that they were not intended to answer fully the large scale impacts of aquaculture in Baynes Sound, but rather to begin to provide some insight into the challenges in trying to address this issue. Two main issues here relate to our lack of understanding of the basic biologies of what are deemed to be important (and in the case of bivalves, commercial) species, and the scales in which investigations need to be conducted to provide meaningful results.

# Description of Baynes Sound intertidal shellfish aquaculture

## *Physical description*

Baynes Sound is about 90 $km^2$ and consists of over 9000 ha of shallow coastal channel fringed by protected bays, open foreshore, tidal estuaries, inshore marshes and adjacent forests. Parts are deeper than 60 m, and the majority of the area is greater than 20 m in depth; its total volume is approximately $7.8 \cdot 10^9$ $m^3$ (B. Carswell, pers. comm.). Comox Harbour, which bounds Baynes Sound on the north, is one of the largest low gradient deltaic deposits on the east coast of Vancouver Island. The shoreline has a great diversity of habitat ranging from hundreds- of-metres-wide intertidal mud and sand flats to rocky shorelines bounding deep water. The surficial geology of the area is predominantly glacial marine, overlain in some areas by fluvial or organic deposits. The unconsolidated sands, gravels and tills dominate most of the beaches except on Denman Island and some of the headlands where exposed bedrock forms a significant portion of the coastline.

Foreshore mapping of the study area (Figure 1) outlines the contrast in the physical shoreline properties between Vancouver Island and the western shore of Denman Island (Howes and Thomson 1983). Vancouver Island is characterised primarily by shore units of beaches, interspersed with low-gradient deltas and tidal flats with nearshore widths extending up to 1000m. The northern tip of Denman Island also has beaches and deltas with nearshore widths up to 500m, but the majority of the western shore is characterised by rock platforms with mixed sand-cobble beach veneer.

The following description of the oceanography of Baynes Sound (except where referenced otherwise) is based primarily on the summary by Morris et al. (1979) of surveys carried out during the 1960s. The primary factors controlling the physical oceanography of the Sound are tides, currents and freshwater. The tides are semi-diurnal, with low waters occurring during daylight or near midnight in the summer and winter months, respectively. The tidal range at the northern end of Baynes Sound is greater than in the south by approximately 0.3 m. On the flood tide, northeasterly currents transport waters from the Strait of Georgia into the northern end of the sound, while the ebb tide is characterised by a greater outflow at the southern entrance. Thus, the net circulation of flow through Baynes Sound is from north to south. Freshwater input is predominantly from the Courtenay River in the north, with smaller streams having only a localised effect (Waldie 1952). The freshwater runoff drives the net outflow of surface waters, superimposed on regular tidal activity with occasional modifications by wind-driven currents. The deepwater currents in Baynes Sound are also presumed to flow towards the south, with a total exchange of bottom water taking place approximately 16 days (Hay & Consultants Inc. 2003).

The waters in Baynes Sound are relatively well protected from wave action by Goose Spit, Denman Island, and the smaller islands extending from the northern tip of Denman Island. This protection helps contribute to the vertical stratification of Baynes Sound waters. There are seasonal variations in density, salinity, temperature and dissolved oxygen coinciding with higher summer temperatures, and inputs of freshwater from heavy winter runoff and spring snowmelt.

### *Protected areas*

Baynes Sound is internationally recognised as important for migratory waterbirds. It has been ranked as the most important wetland complex on Vancouver Island by two of the foremost conservation agencies, the Pacific Estuary Conservation Program (PECP) and the Pacific Coast Joint Venture (PCJV). Conservation values of Baynes Sound have long been recognised. In 1974, to elevate the importance of Baynes Sound as a wildlife area the BC Ministry of Environment [formerly the BC Ministry of Water, Land and Air Protection (WLAP) and before that, the BC Ministry of Environment, Lands and Parks (MELP)], was granted a Notation of Interest Map Reserve over the intertidal foreshore from Maple Guard Point to Buckley Bay. A decade later, international recognition was gained when a series of biophysical studies (led by Environment Canada and MELP) identified Baynes Sound as “critical” habitat for waterfowl.

There are presently five small legislated protected areas (total marine area = 91.7 ha, i.e. <1 km2) within Baynes Sound (Jamieson and Lessard 2000). Provincial policy precludes any new tenuring fronting Provincial Parks. The Rosewall Creek Unit of the Qualicum National Wildlife Area (undetermined marine area) was established by CWS in 1974 for the conservation of essential habitat for migratory birds, and is subject to regulations defined by the *Canada Wildlife Act*. From 1991 to 1996, MELP established Wildlife Reserves at Deep Bay (12.9 ha), Rosewall Creek (Mud Bay) (27 ha), Fanny Bay (51.7 ha), and the Comox/Courtenay River Estuary (undetermined marine area) for the preservation of estuarine habitat and management of waterfowl resources. However, there are no specific provisions under the BC *Land Act* with respect to the management of the Wildlife Reserves (Jamieson and Lessard 2000), and no management plans have been developed for these Wildlife Reserves.

Although not legislated protected areas, there are also two Recreational Shellfish Reserves established by Ministry of Agriculture and Lands (formerly know as Ministry of Agriculture and Food and Fisheries): 14.2 ha of intertidal area and 120 ha of deepwater. These areas preclude shellfish tenures or commercial harvesting. 277 ha of foreshore are held by BC Parks surrounding Sandy Islets Marine Park which is traditionally an area of First Nations clam harvest and continues to be an important area for recreational and First Nation harvest of shellfish.

In British Columbia, local governments have the authority to zone activities on Crown Foreshore. In 1990 the Denman Island Official Community Plan changed the zoning from Aquaculture to Conservation for all areas not under existing tenure to shellfish aquaculture. This new zoning designation does not provide for aquaculture development. Given that Land and Water British Columbia Inc. (LWBC) (Integrated Land Management Bureau (ILMB)) tenure offers are conditional on local government approval requirements, the Conservation Zone now effectively prevents any further development next to Denman Island on a marine area of about 7400 ha (B. Carswell, pers. comm.).

### *Intertidal vegetation*

Intertidal vegetation in Baynes Sound consists of a mixture of red, brown and green algae, with eelgrass beds in the mid-lower zones and marsh vegetation in higher areas. The most important mid-to-lower intertidal vegetation is eelgrass (*Zostera marina, Z. japonicus*), which provides critical habitat for young fish, invertebrates and other species and stabilises shorelines. It also helps to increase water clarity and reduce erosion by reducing wave energy and trapping loose sediments. The areal extent of eelgrass beds (*Zostera* spp.) in Baynes Sound, which includes a substantial admixture of macroalgae, is estimated to be around 500 ha; Comox Harbour is estimated to have an additional 500 ha of primarily eelgrass beds (Romaine et al. 1976, 1981, 1983). In 1995, a more recent assessment from remote sensing data indicated 79 ha of eelgrass within Comox Harbour and 174 ha for the rest of Baynes Sound (B. Carswell, pers. comm.).

### *Wild bivalves*

Intertidal bivalves of Baynes Sound form a rich mixture of native and exotic species, with relative distributions and abundance on each beach determined primarily by the area available at each tidal elevation and the substrate type. Carswell et al. (in press) have shown that within the optimal habitat areas of each clam species (*Protothaca staminea* (native littleneck), *Venerupis philippinarum* (manila clam) and *Nuttalia obscurata* (varnish clam)) within Baynes Sound, no clam net coverage exceeded 20% of a shore type (based on a major substrate) found within each species' optimal habitat areas (with the average below 6%). The epifaunal bivalve community is dominated by two major species groups, mussels (family Mytilidae) and oysters (family Ostreaidae). The infaunal component is dominated by clams of various families, including the Veneridae, Psammobiidae, Myidae, Cardiidae, Mactridae and Tellinidae.

The infaunal community is made up of numerous species, with dominant taxa being determined largely by tidal elevations and substrate characteristics. The bivalve found at the highest elevations is the exotic varnish, or dark mahogany, clam. This species has been recorded from BC since the early 1990s, has quickly expanded its distribution to include the entire Georgia Strait, and is expanding into Puget Sound, Johnstone Strait and the west coast of Vancouver Island (Gillespie et al. 1999). Varnish clams are primarily found at intertidal elevations above other bivalves but overlap with species found lower in the intertidal.

The next zone of the intertidal is dominated by the exotic Manila clam. This species was accidentally introduced to BC with Japanese oyster seed in the

1930s, and subsequently spread throughout Georgia Strait, into Johnstone Strait, up the west coast of Vancouver Island and into the Central Coast to nearly 53ºN (Quayle and Bourne 1972; Bourne 1982; Gillespie and Bourne 2000). Manila clams achieved commercial significance in the late 1980s, and currently are the most important commercial wild-harvest clam species in BC.

The lower intertidal is dominated by the native littleneck clam, found from the mid-intertidal to subtidal depths and of minor importance in commercial fisheries but is targeted, along with manilas, in the steamer clam recreational fishery. Also present are the relatively large, but not now commercially fished, butter clam, *Saxidomus gigantean*, which is harvested recreationally and by First Nations.

## *Salmonids and Pacific Herring*

A minimum of 23 creeks and rivers drain into Baynes Sound, providing spawning and rearing habitat for coho (*Oncorhynchus kisutch*), chum (*O. keta*), chinook (*O. tshawytscha*), pink (*O. gorbuscha*), sockeye (*O. nerka*), coastal cutthroat (*O. clarki*) and steelhead (*O. mykiss*). The intertidal zone of Baynes Sound is utilised as a juvenile rearing area at various times of the year (Healey 1980). Millions of wild salmon juveniles are produced within these watercourses. As well, the Puntledge River hatchery releases approximately 10 million juvenile salmon annually into the Courtenay River estuary and Baynes Sound, including 1.5 million chinook, 3 million pinks, 4.5 million chum and 700,000 coho. The intertidal zone and waters of Baynes Sound are also recognised as productive Pacific herring (*Clupea harengus pallasi*) spawning and nursery habitat on the BC coast (Hay and McCarter 2001). Eggs are deposited on intertidal and subtidal marine vegetation in Baynes Sound and Lambert Channel, and hatched larvae from both areas disperse into the stratified waters of the Sound to rear in the adjacent waters of protected bays and inlets (Haegele and Schweigert 1985; Robinson 1989).

## *Birds*

Baynes Sound – Comox Harbour area is an important staging and wintering area for a wide variety of migratory bird species (Dawe et al. 1998). Designated as an Important Bird Area (IBA), the area includes the Courtenay River estuary to Deep Bay and Mapleguard Point, approximately 35 kilometres to the southeast (Booth 2001). Maximum single day counts recorded during 1980 –1981 surveys found globally significant populations of Pacific Loons, Western Grebes, Brant, Black Turnstones, Mew Gulls, Thayer's Gulls, and Glaucous-winged Gulls (Dawe et al. 1998). The number of bird-use days for the Baynes Sound – Comox Harbour area was highest in winter, second in autumn and spring, and lowest during summer (Dawe et al. 1998). It should be noted that this assessment was after oyster aquaculture, but before clam culture, had been established in much of Baynes Sound, so how it relates to current aquaculture usage by birds is somewhat unknown.

## Oyster and clam aquaculture production in Baynes Sound

The most widely cultured species in the Pacific Northwest is the Pacific oyster. Historically, intertidal production of Pacific oysters was preferred but recent oyster culture trends have been towards deepwater production. Beaches previously used for oyster production are now often used primarily for clam culture (Anon. 1997) if suitable for clams.

The farmed production of clams in BC has been formally licensed only since 1991. Cultured clams have a higher value in comparison to oysters, and the higher quality control associated with culture clams gives them a higher market value than harvested wild clams (Heath 1997). The Baynes Sound area produced $7.6 million of shellfish in 2003, 48% of all farmed shellfish in BC (http://www.agf.gov.bc.ca/fish_stats/aqua-shellfish.htm). Forty-five and almost 100% of the linear shoreline of the west and east sides of Baynes Sound respectively are in presently leased for intertidal shellfish culture (see Figure 1).

There is debate about the utility of the above liner shoreline measures by proponents of clam culture, who note that this is a metric commonly used by opponents of shellfish farming which in the context of habitat and cumulative effects may have little value. Proponents argue that of more relevance is the extent of tenures and the amount of the different habitat types occupied by tenures and clam netting. These data are that shellfish tenures currently occupy 20.3% and clam netting 2.9% of the intertidal area of Baynes Sound as defined by the Baynes Sound Coastal Plan (British Columbia Ministry of Sustainable Resource Management, 2000) (Carswell et al. In press).

Both the above measures of impact are correct, and refer to the spatial pattern of impact. Whether this pattern has significant impacts at a local scale within the Sound, and whether these impacts are acceptable from a cumulative effects perspective is the real issue, and this can really only be addressed by conducting such an assessment.

## Documented effects of intertidal shellfish aquaculture in Baynes Sound

### *Bivalves*

Nets used for clam aquaculture are intended to reduce epibenthic predator pressure on commercial species, and possibly on other non-target species as an unintended result. The addition of clam seed is intended to increase the productivity of the commercial species, which can directly change the population dynamics of this species and possibly change the strength and form of interactions in intertidal communities. In her study, Bendell-Young (SFU, Burnaby, BC, pers. comm.) noted that bivalve density was significantly higher in farm sites, compared to unnetted and unseeded reference sites, but only for one bivalve species, the commercial manila clam, a naturalised exotic species. This species is seeded on to sites, and the increased density observed on sites was close to the level expected due to seeding alone. Since total bivalve biomass did not increase significantly on netted sites, even with the increase in manila clam density, something else was apparently happening. One hypothesis is that predation was not greatly reduced on netted areas since overall bivalve density was similar in both netted and unnetted areas for some species at least, raising the question as to whether netting is effective and worth the cost of potentially alienating habitat from species such as birds, yet alone justifying the dollar cost of netting usage. Other hypotheses are that: 1) within the study sites investigated, clam movement within the substrate may be more extensive that previously recognised, and that at high densities, some clams at least over time may move considerable distances, i.e., to outside the netted areas in this case, in an effort to maintain their densities at some perhaps site-specific level; and 2) because scoters are the primary predators and are preferentially foraging on varnish, then manila, clams, one might not expect changes in densities of other species. Also, although densities of manila clams were higher under nets to the degree that would be expected from seeding alone, this does not necessarily indicate that the nets are not effective at reducing predation. Natural settlement of clams is also likely occurring, and the commercial harvest of manila clams under the nets is high, suggesting that productivity (which is what the producers mostly care about) is higher when nets are deployed. In other words, scoter predation outside nets and human harvest under nets may be roughly equivalent, resulting in similar total bivalve densities under and outside netting. This has not to date been considered when evaluating the efficacy of netting.

Not unexpectedly, Bendell-Young's (SFU, Burnaby, BC, pers. comm.) multivariate community analysis revealed an apparent difference in among-site variability in bivalve composition between farm and reference site groups. Farm sites were slightly more similar to each other, on average, than reference sites, with the loss of 'regional distinctness' among farm sites likely the result of increased consistency in densities of common species at farm sites. The conditions created by clam farming, which are intended to favour the production of commercial species, may create common pressures that drive separate communities toward higher levels of similarity. The homogenizing force of clam farming at large scales appears to be more significant than potential impacts at individual sites. The ability of common farming practices to alter habitat heterogeneity at smaller scales was not documented in the present study, but deserves further research. Nevertheless, the increased similarity among farm sites suggests that impacts of clam farming may be most relevant to larger scale ecosystem processes, such as species' migration, settlement, and resulting meta-population dynamics that may be affected by the spatial structure of habitat and communities. Another plausible interpretation is that farmers are selecting similar beaches for their operations in all three areas, i.e., those sites that have appropriate substrate, exposure, etc., for clam farming. Under that scenario, one would expect farmed beaches to be more similar in bivalve composition, density, etc. Resolving this question begs for a Before – After Control - Impact (BACI) experiment.

Bendell-Young (SFU, Burnaby, BC, pers. comm.) also speculated on the ecological implications for ecosystem processes performed by bivalves if clam farms are increasingly dominated by a single commercially valuable species. It can be argued that if many species contribute to carrying out activities such as filter-feeding, deposit-feeding, burrowing, and nutrient cycling, a monoculture may reduce variability in functional processes, as different species likely operate optimally under different environmental conditions (Yachi and Loreau 1999; McCann 2000; Emmerson et al. 2001). Species-rich communities have been observed to out-perform the best monocultures in total productivity (Emmerson et al. 2001; Tilman et al. 2001). On the other hand, a single species has been suggested to be all that is necessary to carry out a particular function (Worm and Duffy 2003). Species-rich assemblages may simply have a greater chance of including a single, highly active species that results in an overall high level of ecosystem function (Loreau 2000).

In her conclusions, Bendell-Young (SFU, Burnaby, BC, pers. comm.) suggested seeding and netting appear to affect communities at a regional spatial scale, larger than even the largest single site included in her study. If clam farming is a homogenizing force at large scales, then the most important impact of clam aquaculture may be the result of cumulative impacts of several tenures within a given geographical area. Impacts of individual practices remain uncertain, as are the mechanisms underlying many of the results she presented. Given the potential for unknown, large-scale cumulative impacts and the possibility for site-specific responses to farming practices, she recommended that regulatory efforts focus on baseline data-collection, monitoring and site-selection at a regional scale.

### *Scoters*

Coastal BC is the core of the Pacific scoter wintering range and supports globally significant numbers. There are continent-wide concerns about negative population trends with scoters, and thus any factor that is beneficial or detrimental for scoters is of interest (Dan Esler, SFU, Burnaby, BC., pers. comm.). There are overlapping distributions, habitats, and resources between shellfish aquaculture and scoter wintering areas. Some scoter natural history attributes that are relevant for evaluating aquaculture effects are that they spend the majority of year on coastal nonbreeding areas; they feed primarily on bivalves, with Surf scoters (*Melanitta perspicillata*) eating clams and mussels and White-winged scoters (*Melanitta fusca*) eating clams; they feed exclusively by diving for prey (hence only feed in intertidal areas when the tide is in), and they occur in dense aggregations in areas with high prey densities. Wintering scoters might be affected by shellfish aquaculture through modification of food availability, through either increases or decreases in food abundance or accessibility, disturbance associated with industrial activity, or by changes to water quality.

Dan Esler (SFU, Burnaby, BC., pers. comm.) noted that some of the challenges in trying to understand how highly mobile species like scoters are possibly impacted by aquaculture include a lack of historical abundance and distribution data, and temporal variability through the course of a season and between years in the spatial availability of prey and resulting predator distribution (e.g., as areas are grazed down, predators move on). There is the possibility that total clam availability in unnetted areas may actually be increased through movement and reproduction of clams from under the nets. Data desired include how important in general is the Baynes Sound area to scoters within the Strait of Georgia, i.e., what other areas provide high prey abundance; how does prey availability relate to shelter from storms in determining scoter distribution; and what effect has prey abundance change over time influenced scoter distribution (i.e., what are preferred prey species, and how have prey species changed with the introduction and wide establishment at high densities of the recently-established exotic varnish clam.

Recent observations (Dan Esler, SFU, Burnaby, BC., pers. comm.) indicate that there are more scoters now than during the last survey in the winter of 1980-81 (Figures 2,3), this is coincident with both a change in focus of intertidal aquaculture practice (from oysters to clams on many leases) over that period and the establishment at high densities of the exotic varnish clam through the 1990s (which is likely most responsible for observed changes in scoter numbers and distribution), and there is no evidence of negative effects of aquaculture on scoters, i.e., there are no declines in scoter numbers in areas where varnish clam numbers are relatively small. Concerning the latter point, varnish clams make up a much smaller proportion of the clam community in Deep and Mud Bays in Baynes Sound, where aquaculture activities are relatively intensive. This is likely due to their active removal from leases by farmers since they can reach densities which may inhibit the growth of manila and littleneck clams (B. Carswell, pers. comm.). Scoter numbers in those areas increased slightly since 1980-81, roughly in proportion to the prey increase associated with the invasion of the varnish clam. If aquaculture were having effects on scoter densities, one might expect to see the strongest signal in these areas.

Surf Scoter densities were positively associated with the intertidal area, density of varnish clams, a sandy substrate, and the distance to freshwater inflow; there was no significant effect, positive or negative, with any aquaculture attribute. White-winged Scoter densities were positively associated with intertidal area and the density of varnish clams; densities were negatively related to the presence of oyster rafts. However, this is difficult to interpret as the number of rafts present was low, and its not clear why rafts should negatively affect these scoters.

There was a declining survival of White-winged Scoter adults in association with increasing amounts of time spent in areas with shellfish aquaculture (possible negative effects), but juveniles showed the opposite trend. Surf Scoter survival was unrelated to shellfish aquaculture. Results are thus mixed, but indicate no clear advantage or detrimental effect of shellfish aquaculture on scoter survival.

The impact of scoters on intertidal clam abundance where investigated seems to be moderate. There were an estimated >1.5 billion edible size varnish, manila, and littleneck clams along the west side of Baynes Sound (the area along the east side of Baynes Sound (i.e., the Denman Island side) was not considered in this scoter study) in 2003 (based on extensive clam sampling in 2002 and 2003 by SFU and CWS) and scoter consumption estimates indicate that they ate about 11% of this total. However, this analysis has not yet considered clam size removal and available biomass, so actual biomass available and consumption by scoters is not yet known. Contributions of aquaculture to this number are also not clear (i.e., how does seeding, migration, or reproduction of clams under nets contribute to this total?). Varnish clams constitute an important component of Scoter diets, and since these clams were most abundant in Comox Harbour/Baynes Sound area, this correlation is believed to explain why scoter abundance was higher there than in the main part of Baynes Sound, i.e., between Denman Island and Vancouver Island.

Scoters spatial distribution does not seem to be limited by food availability, as indicated by the degree of prey depletion on areas with active shellfish aquaculture. Depletion of clams by scoters was relatively small, indicating that parts of Baynes Sound at least are high quality foraging habitat for scoters and suggesting that shellfish aquaculture is not constraining scoter populations via a mechanism of food limitation. There is thus no evidence at present that current levels of shellfish aquaculture have reduced habitat quality for scoters.

## Summary

Jamieson et al. (2001) documented that intertidal shellfish culture in Baynes Sound is extensive and that much of the area has been modified. They also documented the environmental concerns about the impacts of aquaculture, and the challenges in trying to determine the nature of these impacts in the ecosystem surrounding the farmed areas. While DFO has not undertaken follow-up studies, academics and the Province have initiated some work, briefly summarised here, that demonstrates that determining impacts from aquaculture activities is not a trivial undertaking.

Firstly, the responses of intertidal bivalves to predator control efforts (netting) and stocking with Manila clam seed was not entirely expected, but our understanding of clam biology is not apparently complete, particularly with respect to clam movement potential and how different species respond to bivalve density or biomass. If clams do indeed move tens or even hundreds of metres, then the areal scale of future experimental studies needs to accommodate this potential. Studies evaluating the effectiveness of gear such as predator netting and stocking need to be properly designed to allow an understanding of consequences to be determined, and past study designs may not have been appropriate given this.

The bird study was equally illuminating, in that while some concern has been expressed for the study only looking at the western side of Baynes Sound, the scoter study area included more than 80% of the intertidal area of Baynes Sound, as well as the full range of habitat and aquaculture variation that occurs on the Denman Island side. Regardless, a new understanding of basic bird biology was obtained and the challenges in doing this type of study were clarified. Varnish clams are now recognised as a preferred prey, along with manila clams, and the spatial and temporal variation in seasonal abundance of these clams may largely determine scoter spatial abundance pattern. The absence of prior documentation of scoter spatial distribution in Baynes Sound, i.e. before industrial intertidal aquaculture became extensive and before varnish clams became abundant, makes documentation of impacts now difficult. There are also no historical data to evaluate numerical or distributional changes over time at other study sites in the Strait of Georgia. Thus, it is now only possible to note that scoters seem to have adapted to a changed environment, and that from the work undertaken to date, no conclusive impact from intertidal aquaculture is evident with scoters. It can even be noted that scoters appear to be more abundant coincidently since the advent of clam farming in Baynes Sound, although this is also coincident with the establishment of varnish clams in abundance. It would be interesting to know what scoter spatial distribution might result from an absence of netting, but available food does not seem to be a current limiting factor in determining overall scoter abundance in the Sound.

In summary, recent studies show that the impacts of aquaculture are complex and difficult to determine. Relevant biological understanding of even important exploited (bivalves) and/or aesthetic (bird) species seem to be still limited, and confounding factors such as the abundant establishment of exotic species complicates analyses. Cumulative studies seem to be necessary to try and separate out causal factors, but such studies have yet to be attempted. They will be difficult to undertake and will likely need to be long-term.

The first recommendation of Jamieson et al. (2001) was to establish a collaborative, inter-agency approach to identify both the nature of existing and potential future impacts and, where necessary, to determine how they can be minimised. It was hoped that this would be some sort of cumulative effects assessment, and it this context, the recent review (Duinker and Greig 2006) of the lack of CEA success in Canada in evaluating impacts on valued ecosystem components (VECs) is relevant. Their main conclusions for the failure of CEAs to advance the sustainability of VECs focused around the following issues. 1) a VEC-centered cumulative effects assessment is difficult to do when proponents are focused on getting project approval, and regulators are focused on making sure the impacts of the project are acceptably small. Appropriate data on all other relevant activities is almost impossible to acquire. 2) with the focus on project approval, doing only what is needed to get approval means minimising efforts concerning cumulative effects assessment, 3) effect thresholds for most situations are largely unknown, and the resulting assumption that the system response is linear is often not correct; 4) if a project alone has no negative effect, it is often then assumed that there can be no negative cumulative effect and the cumulative effects assessment is then curtailed, but this assumption is not justified; 5) cumulative effects are often considered a special class of effect that is not considered relevant, and hence poorly considered, when the critical point is simply the need to assess aggregate stresses, and finally, 6) a cumulative effects assessment demands the consideration of alternate impact scenarios, a task that is often trivialised and poorly done. They conclude that the conduct of regionally focused cumulative effects assessments should be the responsibility of governments, and not be a requirement for proponents to undertake. Cumulative effects assessments also need to be followed up by a vigorous monitoring program and subsequent mitigative action as warranted, steps that often fall by the wayside.

Many of the above comments would be relevant to cumulative effects assessments of the sensitivity of habitat for bivalves and scoters in Baynes Sound because of intertidal aquaculture. The circumstance, though, is that for most tenures, a cumulative effects assessment was not required in the first place as most of the tenures in Baynes Sound predate the CEAA. Determining the scale of aquaculture impact that is required to initiate an effective cumulative effects assessment now needs answering. To date, assessment of habitat sensitivity to intertidal aquaculture has apparently had little priority in BC, as Baynes Sound, the main area of intertidal shellfish aquaculture, has yet to be rigorously assessed.

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## CASE STUDY #3 – ACOUSTIC SEABED CLASSIFICATION IN A MUSSEL FARMING BAY (TRACADIE BAY, PEI) (Marc Ouellette)

The objective of this section is to discuss the potential applicability of an acoustic seabed classification system for mapping benthic habitats in relation to their sensitivity to mussel aquaculture activities. This discussion is based on results obtained from an acoustic survey of a suspended mussel culture bay (see also Section 2.4 in the Indicators and Thresholds working paper). Reviews of various acoustic tools and techniques available for the acquisition of benthic habitat mapping data are available in several publications (Kenny et al. 2003; Waddington and Hart 2003; Diaz et al. 2004).

## Introduction

The regions with the most intensive blue mussel culture in Atlantic Canada are predominantly found on the northern and eastern shores of Prince Edward Island (PEI), in the Shippagan Bay and Miramichi Bay of New Brunswick, in the Magdalen Islands and on the southern shore of Gaspésie in Québec, on the southern shore of Nova Scotia and more recently in northern Cape Breton and in northern Newfoundland (Mallet and Myrand 1995; DFO 2002). The methods of culture vary slightly between these regions reflecting particular geographic and environmental conditions such as exposure (wind and waves), water depth, tidal amplitude, hydrological properties and vertical plankton distribution. They all have in common the extreme winter conditions (such as ice cover) typical to our northern latitude. In general, wild mussel spat is collected during the summer, the seed (juveniles) is put into sleeves (socks of various lengths or continuous) in the fall, and subsequent suspended grow-out occurs in the following year. In the optimal areas, harvestable product is available at the end of the second year.

The successful development of mussel culture in Atlantic Canada relative to other bivalve species has been due not simply to chance but rather to biological factors and to certain features of the industry (Mallet and Myrand 1995). The blue mussel (*Mytilus edulis* and *M. trossulus* at low prevalence in the Gulf of St. Lawrence) is an indigenous filter-feeding species that is well-adapted to our temperate environment and capable of tolerating extreme environmental conditions.

### *Mussel aquaculture in Prince Edward Island*

The blue mussel aquaculture industry has been in development in PEI since the early 1980's. Between 1980 and 2001, the mussel landings increased from 40 to almost 18,000 metric tonnes. Production comes mostly from farms along the east side of the Island where the estuaries (and/or bays) tend to be drowned river valleys and along the north shore where the estuaries (and/or bays) are barrier beach lagoons. The mussel farms are primarily found in those regions because these sheltered bays have sufficient water depth (3 to 6 m) for this type of aquaculture. Water temperatures typically range from -2ºC in January to 22-24 ºC in July and August. The salinity usually ranges between 23 to 29 o/oo (DFAE 2003).

## Type of culture

The suspended longline system (buoyed backline connected to anchors, at either end, by means of scope lines) is the culture method utilized throughout the province. Mussel seed is usually collected in the upper reaches of inlets or rivers where shallow water depths limit grow-out operations. The mussel larvae settle on rope collectors and grow rapidly, reaching sizes of 10 to 25 mm by fall. Harvesting of the seed occurs between early October and late November. The seed is manually stripped from each collector, graded and then loaded at a density of 120 to 240 mussel seeds per foot (400 to 800 mussel seeds per meter) into a mesh tube (sleeves or socks). Individual socks are about 40 mm in diameter and average 2.5 to 3 m in length depending on local water depths. These socks are then attached and evenly spaced along the buoyed backline. The length of the longline typically varies from 80 m to 100 m but may be as long as 150 m. The mussel seed migrate through the mesh and become attached to the outside of the sock by byssal threads with their siphons pointed outward. The first winter and the following year is the grow-out period.

Longlines are completely submerged during the winter because the bays in PEI freeze-over with normal ice thickness between 30 and 90 cm. In the spring some growers refloat their longlines to the surface in order to take advantage of the warm, often more productive waters, and to examine them for predators and/or fouling to be removed. One method of removing fouling organisms, such as the second set of mussel spat, is to temporarily (7 to 10 days) lower the lines so that the mussel socks touch the sea floor. Rock crabs and/or starfish are then able to climb onto the mussel socks and remove the second set of smaller mussel spat and other epifauna. Another method is pressure washing the socks with seawater. Some growers prefer to leave the mussels sunk for the entire grow-out period. In 18 to 24 months, the mussels reach a marketable size of 55 to 60 mm. Harvesting is conducted through ice covered bays in winter, with specialized techniques, or in open water during spring for a maximum meat yield and a good shelf life (Mallet and Myrand 1995; DFAE 2003).

## Environmental interactions

The positive and negative effects of marine shellfish aquaculture on fish habitat will be addressed by authors, among others, in the first working paper (Terms of Reference for the National Peer-Review Workshop). Furthermore, the diverse influences that populations of suspension-feeding bivalves exert on marine ecosystem processes have comprehensively been reviewed by Dame (1996), and Dame and Olenin (2005). However, in the effort to answer the question of “what types of fish habitat are likely to be affected by shellfish aquaculture, and what is their relative sensitivity?", it is important to review some of the key influences that bivalve species can exert on their environment.

Bivalves serve as key agents in benthic-pelagic coupling because they feed on seston and transfer undigested organic and inorganic material in their mucus-bound feces and pseudofeces (biodepositions), which sink to the sediment surface (Dame and Olenin 2005). The complex relationships among some benthic and pelagic processes that may be influenced by benthic bivalve suspension-feeders are partially summarized in Figure 4. The biodepositions can be extremely important in regulating water column processes where bivalves are abundant in coastal waters and in seasons when water temperatures are warm enough to promote active feeding. The feeding response of bivalves to changes in seston concentrations varies considerably among species. The species that can exert the greatest influence on benthic-pelagic coupling are those that maintain high clearance rates (such as the eastern oyster, *Crassostrea virginica*, and the blue mussel), even when seston concentrations increase, and reject large numbers of particles as pseudofeces (Newell 2004). In addition to the direct "top-down" grazer control that bivalves can exert on phytoplankton stocks, they may also exert "bottom-up" control by changing rates and processes of nutrient regeneration. Nitrogen and phosphorus, excreted by the bivalves and regenerated from their biodeposits, are recycled back to the water column and support further phytoplankton production (Newell 2004).

The sediment-water interface is one of the most important transition zones for solute exchange; it is characterised by steep gradients and extensive spatial and temporal heterogeneity. As diagenetic reactions in surface sediments are dramatically affected by biogenic activity, decomposition of sedimentary organic matter and nutrient remineralisation can be rapidly accelerated within the biogenic mixing zone. This results in a vertical colour transition from brown at the sediment surface to olive black at depth. Colouration is dictated by the redox state (ferrous or ferric) of the dominant electron acceptor, iron, and is photogenically distinct: oxidised sediment appears more reflective than the underlying reduced sediment (Solan et al. 2003).

The diagram (Figure 4) also shows the indirect influences of bivalve feeding on surrounding habitats in reducing turbidity, by the removal of both phytoplankton and inorganic particles from the water column. The resulting increased light penetration to the sediment surface can potentially enhance the production of ecologically important benthic plants, such as seagrass (Newell and Koch 2004). Increased water clarity also promotes the growth of microphytobenthos (MPB). These benthic algae are an important food source for both sessile and mobile benthic herbivorous meiofauna and macrofauna that, in turn, are eaten by many carnivorous fish. Consequently, an abundant MPB community can support higher trophic levels (Miller et al. 1996).

Mussel socks and associated aquaculture structures in the water column, such as ropes and buoys, provide an ideal surface for the settlement of various species (considered as a fouling community). Foulers are composed of filter feeders (competitors), herbivores, detritivores, and/or predators. This fouling growth serves as a food source for many animals and to some extent provides the type of spatially complex habitat that is sought by many species of animals (Newell 2004). These fouling species can also exert influences on the benthic-pelagic coupling and surrounding habitats. The assemblage of this fouling community also varies over time (seasons) and with fluctuating environmental conditions (LeBlanc et al. 2003). Therefore mussel socks, in addition to its fouling community, can have different influences on the ecosystem at different times. Considering that mussel socks will spend at lease 12 to 14 months in the water column, they could truly be considered as living reefs. Ellis et al. (2002) identified 32 species, distributed in 6 Phylum, which were found on or associated with mussel socks in PEI waters. Unfortunately, more recently we have found aquatic invasive species that can also be associated with fouling communities on mussel socks in various bays of PEI; a green algae (*Codium fragile*) (MacNair 2002), the green crab (*Carcinus maenas*) (Audet et al. 2003), and filter-feeders such as the clubbed tunicate (*Styela clava*), the vase tunicate (*Ciona intestinalis*), the golden star tunicate (*Botryllus schlosseri*) and the violet tunicate (*Botrylloides violaceus*) (MacNair 2005).

This brief discussion on environment interactions is an attempt to show that the types of fish habitats that are likely to be influenced by shellfish aquaculture in shallow-water estuaries are numerous (e.g. epibenthic and endobenthic habitats, submerged aquatic vegetation habitats, fouling communities associated with mussel culture, plankton communities, …). Bivalve species, indigenous and cultivated, are an integral component of coastal ecosystems and, coupled with hydrodynamic processes, can have both direct and indirect effects on various other biotic communities.

## Habitat sensitivity & cumulative effects

Studies carried out on the impact of shellfish farming on the benthic environment present various data sets that suggest a large spectrum of effects ranging from small to important. This wide range of impacts observed in the literature is largely related to various local effects such as the heterogeneity of the coastline, various oceanographic and biological parameters as well as husbandry practices (Miron et al. 2005). The type and intensity (scale) of the culture activities, the seasonal and physical characteristics of the aquaculture site (e.g. water depth and circulation, type of substrates, sediment porosity, type of benthic biotic communities), and the state of the marine habitat being assessed, in relation to other anthropogenic activities, are all determining factors in terms of habitat sensitivity to shellfish aquaculture.

Coastal waters worldwide are increasingly enriched with nitrogen and phosphorus as a consequence of agricultural fertilizer run-off and sewage inputs from growing human populations along coastal margins. This anthropogenic nutrient enrichment is causing fundamental changes in the patterns and magnitude of primary production, including linked changes in: water turbidity, distribution of vascular plants and biomass of macroalgae, sediment biogeochemistry and nutrient cycling, nutrient ratios and their regulation of phytoplankton community composition, frequency of toxic/harmful algal blooms, habitat quality for metazoans, reproduction/growth/survival of pelagic and benthic invertebrates, and subtle changes such as shifts in the seasonality of ecosystem functions (Cloern 2001). Coastal eutrophication has been identified as an important ecological problem in many regions of PEI, especially on the north shore where tidal amplitude is weaker than on the south shore. Concentrations of total nitrogen, nitrates, and phosphorous have increased since the 1970s. The ranking of concentrations of these nutrients corresponds with the proportion of the watershed that is used for agriculture (Raymond et al. 2002). Sedimentation from land activities and water turbidity also contributes to this ecological problem (Meeuwig et al. 1998). Eutrophication can effect the ratios of pelagic to benthic primary production, oxygen to sulphate respiration, and proteolytic to carbohydrate decomposing enzyme activities. The structure and function of microbial biofilms colonizing stones and sediments can also reflect a change in trophic status (Meyer-Reil and Köster 2000).

Natural and aquaculture-reared stocks of bivalves are potentially a useful supplement to watershed management activities intended to reduce phytoplankton production by curbing anthropogenic N and P inputs to eutrophied aquatic systems.

In addition to their important role in the mechanism of nutrient cycling, there is a substantial amount of N and P, absorbed by bivalves during tissue and shell growth, which is directly extracted from the system by harvesting (Rice 2001; Landry 2002; Newell 2004).

However, biodeposition at very high bivalve densities, coupled with high food availability and poor water circulation, can be intense enough that the resulting microbial respiration reduces the oxygen content of the surrounding sediments and therefore have an effect on the benthos. Furthermore, seasonal and physical characteristics of the aquaculture site, such as sediment porosity, water flow, abundance of bioturbators, and so forth, can directly and indirectly influence sediment biogeochemical processes (Newell 2004). The magnitude of these possible variations is illustrated in a study by Sundbäck et al. (2000) of two different grain size sediments. They reported that coupled nitrification-denitrification rates were about an order of magnitude higher annually in finer grain sediments with active bioturbators than in sediments of higher porosity and with a slightly lower biomass of bioturbators.

Grant et al. (1995) found that although sedimentation rate was higher under mussel culture lines than at an adjacent reference site of similar sediment texture, in a south shore Nova Scotia farming cove, the impact on the benthos appeared to be minor. They suggested that mussel fall-off and other anthropogenic effects (e.g. causeway) probably had more impact on the overall benthic assemblages than the result of biodeposition. Tita et al. (2004) evaluated the environmental characteristics (such as sediments, benthic macrofauna and meiofauna composition, and oceanographic data) of a newly developed suspended mussel culture site in the Baie de Plaisance (Magdalen Island, Québec). It will be of interest to follow the temporal and spatial changes of this site in order to increase our knowledge on the influence of shellfish aquaculture on the benthic assemblages.

The documented literature on the effects of suspension mussel culture on sensitive habitats in temperate waters is somewhat limited at the present time. However, there are several interesting papers from other regions of the world that can offer insights on potential influences of blue mussel culture that could be encountered here. Mirto et al. (2000) studied the impact of organic loads due to the biodeposition of a blue mussel farm in a coastal area of the Tyrrhenian Sea (Western Mediterranean). They found that densities of microbial assemblages beneath the mussel cultures increased and displayed, when compared to the control, a larger cyanobacterial importance associated to a strong decrease of the picoeukaryotic cell density. Farm sediments displayed significant changes in meiofaunal density: turbellarian (flatworms), ostracod (crustaceans) and kinorhynch (marine microscopic worms) densities decreased significantly, while copepod densities remained constant or increased possibly profiting from the enrichment in microphytobenthic biomass associated to mussel biodeposits. However, the comparative analysis of the mussel biodeposition and fish-farm impact on sediments beneath the cultures revealed that mussel farms induced a considerably lower disturbance on the benthic community structure.

Hartstein and Rowden (2004) found a relationship between the hydrodynamic regime of a farm site in New Zealand, organic enrichment of seabed sediments by mussel biodeposits, and a subsequent modification of the macroinvertebrate assemblages. Multivariate analysis revealed that there were significant differences in macroinvertebrate assemblage composition (averaged across seasons) between samples taken inside and outside of the two relatively low energy sites, whilst no such difference was observed for the relatively high energy site. Chamberlain et al. (2001), in southwest Ireland, also found that variations in the dispersion of biodeposits caused by local current patterns had a significant influence on the impact observed between suspended mussel culture sites.

Crawford et al. (2003) also found that sediment deposition, redox values, sediment sulphide concentrations, organic carbon content and water turbidity levels near the bottom were significantly different between suspended culture farms, in Tasmania (Australia), but not between sites outside the farm and sites within the farm. The benthic infauna did not show clear signs of organic enrichment, and neither univariate nor multivariate measures of benthic infauna were significantly different between sites inside and outside the farm, although they were different between farms.

Baudinet et al. (1990) found that although the flow of nutrients towards the water column is higher in mussel farming zones than in other areas, biodeposit input into the sediment under mussel ropes did not affect the equilibrium of the ecosystem in Carteau Cove (France) when the results where put in a seasonal and spatial context.

Crawford (2003) found that a qualitative risk assessment of detrimental impacts of shellfish farming in Tasmania (Australia), including the blue mussel, rated the risk of spread of introduced pests and/or pathogens as high. However, this high risk rating also applies to many other activities in the marine environment, such as commercial and recreational fishing and sea transport. The level of risk due to habitat disturbance was rated as moderate within the lease area, but it is not expected to extend outside the farm. Risks of organic enrichment of the seabed and reduced food resources for other filter feeders were both rated as low.

Disturbance effects that alter biotic interactions must be considered when assessing impact because these interactions are important in shaping community dynamics. This argues for a more thorough understanding of the factors regulating abundance and species composition in natural benthic communities. The criterion as to what constitutes an impact is extremely subjective, and dependent on user criteria for acceptability (Grant et al. 1995).

In all cases, because of cumulative environment effects related to various anthropogenic activities in bays and estuaries, tools and methodologies that could assist in evaluating the state of bay-scale parameters and marine habitats are needed. The bay-scale data is particularly important for a proper interpretation of farm-scale measurements in the assessment of shellfish aquaculture activities. Emphasis needs to be directed at understanding the complexities of coastal system functions rather than simplifying and scaling down the system into smaller components (Diaz et al. 2004). An inadequate interpretation of the actual effects may fuel the agenda of opposing parts while assessing the ecological impact of shellfish farming on the benthic environment (Miron et al. 2005).

## Acoustic seabed classification

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

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

Benthic habitat mapping is a multidisciplinary task that combines physical (geological), biological, oceanographic, and chemical components of the seafloor. Data such as substrate type, topography, biological species, and oxygen concentration are all necessary to create an accurate picture of a habitat

(Diaz et al. 2004). The acquisition of benthic habitat data is typically a costly and time-consuming effort. However, advances over the last decade in technologies and disciplines associated with the field of geomatics show a high potential for the acquisition and analysis of some of the data layers needed in benthic habitat mapping. Geophysical techniques that help identify and define large-scale marine benthic features are valuable in appraising essential habitats of marine benthic assemblages. Most importantly, these technologies are capable of providing accurate and repeatable measurements. These are critical requirements for measuring spatial and temporal variations of the seabed that could be associated with anthropogenic activities.

The acoustic method (single beam sonar) of remote sensing data collection is an accurate, low-cost, and relatively simple technique for generating seafloor topography and characterization of the surface sediment composition (with acoustic seabed classification systems), especially in areas with gradual seafloor relief or shallow water depths. These acoustic techniques can operate over a wide frequency band, from less than 1 kHz for some of the lower-frequency sub-bottom profiling systems to over 1000 kHz for some of the higher-frequency side-scan sonar systems. Sub-bottom profilers (very low frequencies) provide high-resolution definition of sediments down to a maximum of about 50 m in soft sediment and much less in coarser sediments or in shallow water. These devices offer the potential to map sediment thickness, infaunal communities and to examine interactions between benthic fauna and sediments (Kenny et al. 2003). In all acoustic systems, an increase in frequency leads to an increase in resolution and a decrease in range or depth of coverage (Waddington and Hart 2003). Given their various configurations, acoustic systems should be selected in accordance to the primary objective of the type of benthic habitat being mapped (e.g., higher frequencies for submerged aquatic vegetation (SAV) mapping, medium frequencies for epibenthic mapping and lower frequencies for endobenthic mapping).

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

### *Habitat characterization*

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

The key to successful application of this technology, however, lies in the translation of basic physical data on bottom substrate and characteristics into meaningful representations of benthic habitat quality (Diaz et al. 2004). The characterization or classification of benthic habitats may be based on a wide variety of seafloor (topography, composition, complexity) and water column (salinity, temperature, turbidity, dissolved oxygen) physical parameters, as well as the actual observed biological structure. The physical characterization of the seafloor is undoubtedly one of the most important elements in any comprehensive benthic habitat classification scheme (Waddington and Hart 2003).

## Case study: acoustic survey in Tracadie Bay, PEI.

### *Introduction*

Tracadie Bay is a shallow (mean of 3 m with a maximum of 6 m water depth), nearly enclosed tidal lagoon (surface area of 14 km2) located on the north shore of PEI. The general oceanographic characteristics of this bay are shared by a number of other bays on the north shore of PEI and the Gulf coast of New Brunswick. The overall Tracadie Bay system is made up of Tracadie Bay proper as well as the adjoining Winter Bay, to the west, which is connected by a relatively narrow constriction (Figures 5 & 6). The principal freshwater input into this system is from Winter River situated at the head of Winter Bay. The Tracadie Bay system is ice covered in the winter months (Dowd et al. 2001).

The Tracadie Bay system is an important area for shellfish aquaculture with a major development focus on the blue mussel. Tracadie Bay is principally used for the grow-out phase of the production cycle whereas mussel spat collection, and early juvenile growth, is mostly done in Winter Bay. On a much smaller scale, there is also the presence of some culture sites for the eastern oyster

(*Crassostrea virginica*) in the south-eastern area and the soft-shell clam (*Mya arenaria*) in the north-western area (DFO 2002).

A research project was initiated in 2002 in order to evaluate the distribution and structure of wild mussel populations and the importance of cultured mussel fall-off on the sea floor. This was part of a larger ESSRF funded collaborative project on “Integrated Ecosystem Studies for Modelling Mussel Aquaculture - Environment Interactions”. A secondary objective of this study was to evaluate the application of an acoustic seabed classification system in a mussel culture bay. One of the challenges in using this kind of technology, in a shellfish aquaculture setting, is to avoid and/or eliminate possible acoustic interferences from the water column (such as mussel socks, cables and buoys). One important component of this objective is also to conduct benthic habitat characterisation of the resulting acoustic seabed classification map.

## *Materiel and methods*

The acoustic seabed classification system that was used in this study was a QTC View-V, specifically designed for shallow water mapping. The echo sounder is a single beam Suzuki ES2025 with a transducer frequency of 50 kHz and a beam width of 24º (Preston and Collins 2000). The vessel speed during the survey was 5 to 10 km/h depending on navigational hazards and the weather. The survey track lines were spaced approximately 500 m apart. A first survey was conducted in late August 2002 and a second in September 2003. The second survey track lines were conducted in the middle of the ones done in 2002 in order to increase coverage (250 m spacing) of the bay. Post-processing of the data, using the waveforms editor in the QTC Impact software (Version 3.4, Quester Tangent Corp.), allowed us to identify and eliminate water column interferences from the dataset before conducting the seabed classification analysis (QTC 2004). Ultimately, the dataset was exported to GIS software (MapInfo 7.0, MapInfo Corp.) for additional analysis and various interpolations of the data.

The approach of this system to seabed classification involves three steps: echo digitization of the first returning echo during data acquisition; echo description by the application of a set of algorithms to analyse and generate a series of features; and echo classification where the most useful features are chosen by principal components analysis and assigned an acoustically distinct class representing the seabed (QTC 2004).

## *Results and discussion*

This type of system was shown to be efficient for mapping bathymetry (0.1 m resolution) and certain substrate features in a mussel culture bay. The acoustic seabed classification map (Figure 5), generated from the analysis of this acoustic dataset, shows the spatial coverage of four acoustically distinct types of seabed. Furthermore, the acoustic seabed classification results were consistent between the two surveyed years, observed in similar class areas and at cross-tracks, which demonstrate the capacity of this system to provide accurate and repeatable measurements. This approach is therefore well suited for the measurement of spatial and temporal variations (monitoring) in some physical characteristics of the seafloor.

Preliminary results in the habitat characterization efforts of the various seabed classes show that one class (red) is generally associated with flat bottoms of very soft mud with very little or no SAV and very little epibenthic fauna. A second class (yellow) is also associated with relatively flat bottoms, but with more consolidated mud, the presence of some SAV (not always) and more benthic fauna (including the presence of bacterial mats in some areas). The remaining classes (green and blue) are associated with more complex benthic habitats (bottom not always flat, sand-mud substrates, the presence of denser SAV (not always), and the presence of endobenthic and epibenthic fauna). The similarities between classes can also be observed in the first principal component (Q1) interpolation map (Figure 6) representing the dominant seabed acoustic features. The red color (Q1 value) seems to be associated with homogenous substrates (flat and soft) whereas the blue-cyan colors are associated with more heterogeneous substrates. These results compare fairly well with a physical characteristics map, generated by MacWilliams (1974), which qualitatively looked at the consistency (type and compactness) of bottom substrates. The seabed differences could partially be explained by the topography (which can influence SAV coverage) and water circulation (which can influence sedimentation patterns) of this bay. The first and second classes are generally associated with the deeper part of the bay whereas the others are associated with the shallower and more dynamic shoreline. The general tidal circulation pattern of Tracadie Bay indicates that the central region of the bay has relatively strong tidal currents while other regions, particularly the southern end of the bay, have relatively weak currents (Dowd et al. 2001). There is also the wind driven effects, during the ice free seasons, which can influence the patterns of sediment deposition and resuspension.

The type of substrate (sediment grain size) and the state of that substrate (such as shear strength and porosity) have a major effect on echo penetration. The substrate topography (such as flat bottom, sand ripples, rocks and slopes) and the reflectivity of that material have an effect on echo scattering (Preston et al. 1999; von Szalay and McConnaughey 2002; Kenny et al. 2003). The biotic community (endobenthic and epibenthic) and patchiness can also have a major influence on the acoustic signal. In theory, the structures (e.g. shells) of benthic fauna could influence the acoustic reading if the size and/or densities of individuals are great enough. The burrowing activities of several species could also have a significant influence on the state of the substrate (Gray 1981). For example, where infaunal invertebrate species have particularly strong impacts on sediment structure (bioturbation), acoustic methods could prove useful in locating nursery grounds and habitats containing large species, but provide little assistance in understanding fine-scale species interactions or identifying the factors controlling assemblage structure (Solan et al. 2003).

The type and densities of SAV (such as eelgrass beds), caused by the acoustical reflectivity of the gas-filled plant stems and/or blades, and biogenic accumulations (such as bivalve reefs) can also influence echo scattering (Sabol and Johnston 2001). This complex assemblage of physical and biotic communities, along with the chemical particularities, constitutes a major challenge in benthic habitat characterisation and further research is needed in order to properly interpret this acoustic seabed classification map. Several research projects have found similar results in that these acoustic seabed classification systems can provide very valuable data for benthic habitat mapping, such as topography and general differences in substrate. However, because of acoustic mismatch between physical and biological attributes, careful ground-truthing is required to ensure that the acoustic class splits are biologically relevant (Morrison et al. 2001; Ellingsen et al. 2002; Freitas et al. 2003; Pinn and Robertson 2003).

There is no obvious relationship between the distribution of the acoustic habitat classes and the location of mussel culture leases in this bay. However, a thematic map showing surveyed aquaculture leases (polygons) can be misleading when considering the intensity of mussel culture in a bay. The husbandry practices associated with the production cycle, from mussel seed to harvestable product, can be highly variable between cultured leases because not all are utilised in the same way at a given time. Several factors need to be considered in assessing the potential environment effects of a cultured lease, such as stocking density (sock mussel densities, length of socks, spacing between socks, number of socks per longline, length of the longline, spacing between longlines and number of longlines per lease) and orientation of the longlines in relation to water circulation. Also, some leases will have new mussel seed (fall), juveniles, market mussels and/or (lease can be subdivided by farmer) be empty. In addition, some farmers double sock to prevent mussel fall-off and/or try to control the epifauna while others don't.

Furthermore, the depth (volume) of the acoustic measurement in the substrate at low sonar frequencies such as with 50 kHz is mainly subsurface (several centimetres), depending on the type and state of the sediments (Collins and Rhynas 1998; Preston and Collins 2000). This suggests that the physical changes of the substrate that could be associated with mussel culture leases would be at a more superficial layer. This is something we plan to investigate now that the system being used is also equipped with a higher frequency (200 kHz), which in theory should be capable of generating a more superficial acoustic map of the seabed. Nonetheless, a study by Shaw (1998) also showed no significant differences between lease and reference sites within cultured bays. He collected sediment core and Ekman grab samples in 20 estuaries throughout PEI, including Tracadie Bay. Core samples were analysed for water content, organic content, redox potential (Eh) and total sulfide levels. Grab samples were analysed for biotic information.

Miron et al. (2005) did not find any conclusive differences between cultured sites for sediments (granulometry, organic matter, sulphide content and Eh) and macroinvertebrate diversity in the Tracadie Bay system in relation with husbandry practices. BIOENV analyses showed that culture density explained a small proportion of the benthic assemblages' variability underneath mussel lines when using the macroinvertebrate abundance data set. Similar analyses showed that water depth better explained the variability observed under mussel lines when using the macroinvertebrate presence/absence data set. The absence of a strong relationship between husbandry practices and the studied benthic parameters might be related to the oceanographic characteristics and land-based activities associated with the water system rather than direct and cumulative effects of mussel culture (Miron et al. 2005).

## Summary and conclusions

An acoustic remote sensing system can efficiently be used in collecting bathymetry data, which can be interpolated to generate a continuous topographic map of the seafloor. This layer of information is crucial in any comprehensive benthic habitat mapping project. A single beam sonar system has proven to be useful in mapping relatively shallow bays and estuaries, including bays with extensive mussel culture activities. This system, when interfaced with acoustic seabed classification coprocessors, can analyse the returning echo for various features of the seabed, such as substrate hardness and roughness. This data can then be used in the characterization of the substrate composition of the seabed, another important mapping layer in identifying and delimitating benthic habitats. Furthermore, the system used in this study was able to generate accurate and repeatable measurements of the seabed during the two surveyed years. This suggests that this tool could be used to assist in monitoring spatial and temporal changes associated with the acoustic physical characteristics of the seabed. There was no obvious relationship between the acoustic data obtained in Tracadie Bay and the location of mussel culture leases in this bay.

Additional research is needed in the characterizations of benthic habitat based on acoustic measurements. This remains a challenge, mostly because the seabed parameters that can influence the acoustic signal are numerous, complex and variable. The link between the acoustic waveform analysis and the seafloor classification scheme must be based upon extensive ground-truth data that are acquired over the different seafloor types likely to be encountered. The problem of data density mismatch between physical and biological methods will likely not be solved until acoustic methods can routinely resolve the elusive biological components that make a physical substrate a habitat (Diaz et al. 2004). This is required to ensure that the acoustic class splits are biologically relevant.

Diaz et al. (2004) listed several primary conditions to be met in order to proceed with the development of a fully integrated marine classification scheme that can meaningfully resolve habitat while also allowing interrogative assessments of ecosystem sustainability and integrity to be undertaken:

1. target organisms or other living resources requiring protection or management must be specified because a habitat deemed of high quality for a particular group of macroinvertebrates, for example, may not necessarily be of similar high value for fish or other resources,

2. for these organisms it is most important that appropriate biological interpretations be applied to the classification of physical substrate types, so that biologically/ecologically meaningful habitats are mapped rather than simply substrate type,
3. selected metrics or indices for assessing organism–substrate relationships should be based on ecological principles, their use be a priori justified and validated and they must be highly correlated to target organism life histories and habitat requirements,
4. the results of any habitat mapping or assessment should be interpreted within a ecosystem-wide framework, and
5. it must be recognized that an interdisciplinary approach is needed when combining these components into a benthic habitat classification and quality methodology.

Finally, acoustic seabed classification systems presently available on the international market by several companies have different approaches on how they conduct the acquisition and analyses of the single beam acoustic data. The established relationship between the acoustic waveform and the seafloor type is very dependent on both the echo sounder settings (e.g., frequency, power and gain), the systems calibration and seafloor type. This relationship would need to be re-established for each new project area, or anytime the echo sounder settings have been modified (Waddington and Hart 2003). Thus, data sharing is presently hampered by the lack of uniformity and standards in data collection, classification and processing protocols (Preston and Collins 2000; Kenny et al. 2003; Diaz et al. 2004). Owing to the importance of seafloor characterization across numerous nationally important applications, there is a great deal of government-funded research and development underway to improve various aspects of the acquisition and interpretation of seafloor characterization data (Waddington and Hart 2003).

## GENERAL SUMMARY

The three case studies presented were originally intended to provide some information on the relative sensitivity of different marine habitats to shellfish aquaculture. However, the extent and quality of research studies on this topic is relatively poor at this point – all three case studies refer to recent work which has not yet been published, and a considerable proportion of shellfish aquaculture impact information appearing only in the grey literature (i.e. not peer reviewed). Jamieson et al. (2001) still applies, as paraphrased in case study #2:

> "The literature is fragmented in its relevance, and much available information has not been scientifically reviewed and published. Views expressed are thus more hypothesis-generating than definitive, which warrants a need for rigorous testing and evaluation."

Thus we cannot quantify the relative sensitivity of different habitats to shellfish aquaculture, or provide an opinion on any regional differences (e.g. west coast versus east) which may or may not exist. Our efforts were not completely fruitless, however, and it is important to list some observations here:

- Eelgrass remains an important and sensitive marine habitat (Vandermeulen 2005), and shellfish aquaculture within eelgrass beds should be closely assessed and monitored.
- Although intertidal soft bottoms are physically relatively harsh environments (movement of sediment, desiccation at low tide, exposure to terrestrial and aquatic predators, etc.), they are also spatially limited in a vertical and horizontal dimension. Organisms living in this zone are therefore sensitive to anything which would crowd them out of this limited space. There is a growing list of aquatic invasive species (AIS) which target shallow or intertidal bottoms, and their potential to crowd out native species has been demonstrated in more than one instance. One must bear in mind that most cultured bivalves in Canada are not native to the areas where they are placed for grow out (or in some cases not even native to Canada, they are true AIS) – and the sensitivity of native species to displacement must be considered.
- The physical characteristics of the water column itself can lead to a level of habitat sensitivity in the face of shellfish aquaculture. A shallow, protected (i.e. low wave action or current speeds), well stratified, soft bottomed bay with warm summer surface temperatures and abundant phytoplankton may not be able to maintain an oxygenated surface sediment layer (or even an oxygenated water column) when an additional particulate carbon supply is added to the sediments by way of bivalve pseudofaeces at high aquaculture densities.

Due to the same lack of quality information mentioned above, it is also not possible to make a statement regarding the relative effects of different aquaculture systems on sensitive habitats. Other than avoiding aggressive harvest methods like whole scale dredging and driving big trucks in the intertidal, most bivalve aquaculture impacts seem to be related to the scale of aquaculture infrastructure involved rather than the type of infrastructure used.

Since impacts on sensitive habitat appear to be scale related, the quantification of effects and management regime needs to be revisited. A framework to accomplish this is presented in the next section.

# IMPLICATIONS FOR MANAGEMENT

The impacts of shellfish aquaculture on sensitive marine habitats are scale related and cumulative. At low density, shellfish aquaculture impacts may be only ‘footprint’ related and affect all or only a portion of the benthic sensitive habitat (e.g. eelgrass or native bivalves) located directly under the aquaculture site. At higher densities, the individual footprints may begin to coalesce and completely displace a sensitive benthic habitat feature like an entire eelgrass bed or a unique and sensitive infaunal assemblage. Once bay scale bivalve aquaculture is attained, the sensitive habitats within the entire ecosystem at that scale may be affected - most obviously by direct displacement due to the aquaculture infrastructure and sheer biomass of cultivated organisms, but also indirectly by alterations in predator abundance, water quality, filtration of food supply and propagules, and sediment geochemistry.

We suggest a bay scale management approach, even if bivalve aquaculture has not yet attained that scale on site[6]. Bay wide management allows for the selection of reference sites (or even protected sites) to gauge impacts and protect sensitive habitats, and focuses monitoring methods at an appropriate scale to capture cumulative impacts. Bay wide management is also the only way to control the pace of aquaculture development and determine when the carrying capacity has been reached – it is a true opportunity for adaptive management that is informed by monitoring as a feedback loop. The advantage to industry with this framework is a more informed (and therefore more predictable) management regime and a built in estimate of carrying capacity which allows for economically viable industry expansion, rather than overdevelopment and subsequent deterioration of bivalve growth rates or collapse.

Bay wide management also places shellfish aquaculture within the context of other human activities in the system, including watershed activities that may affect the bay. Multiple human activities in the water and on land cumulatively alter bays, and shellfish aquaculture is another cumulative factor which may positively or negatively affect the final ecosystem health of the bay. Bivalves at density are ‘ecosystem engineers’ (i.e. they can have a major impact on material

[6] This is a scale of 10s of kilometers, rather than 1 km or 100s.

and energy flow). As such, they are uniquely coupled to their ecosystem - bivalves are part of and interact with other cumulative effects.

Science will be challenged by the bay wide / adaptive management approach. New monitoring methods will need to be developed (as in case study #3) which operate at different scales while offering cost effective indicators that can trigger management action. Sensitive habitats and ecosystem level parameters will both require tracking. As the case studies note - research is already being proposed along these lines, all that is required is the commitment to support the work.

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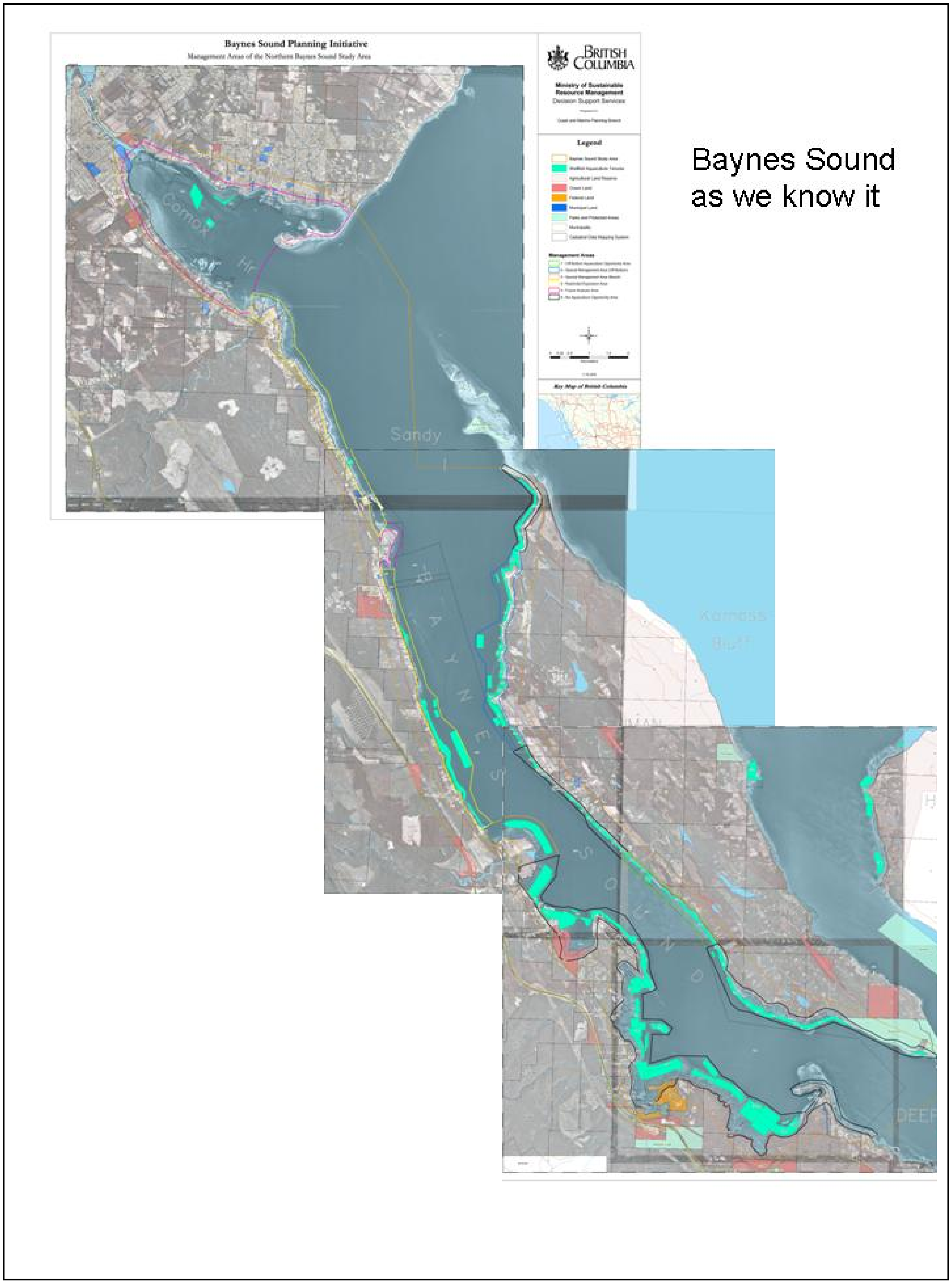


Figure 1: Baynes Sound and Comox Harbour. The blue-green polygons are 2003 intertidal shellfish leases.

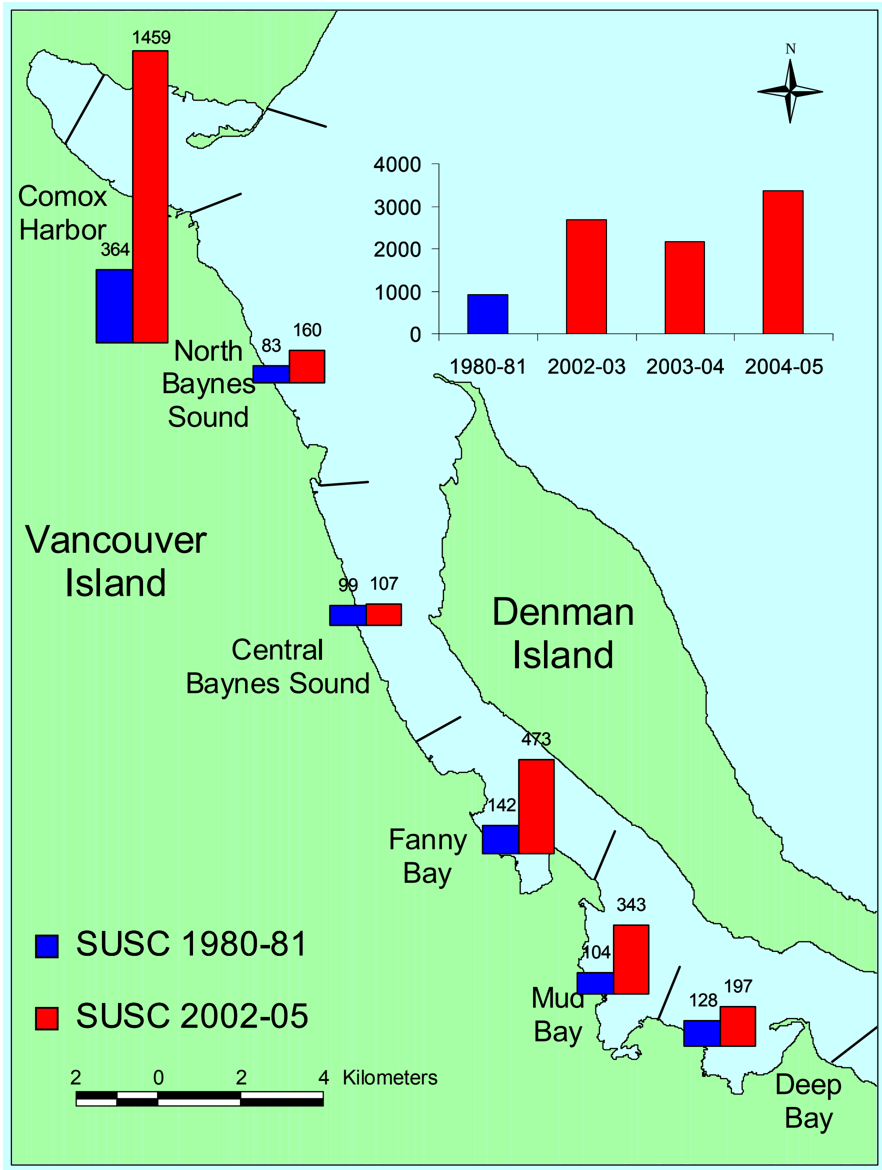


Figure 2: Surf scoter abundance on the Vancouver Island side of Baynes Sound, SUSC = surf scoter.

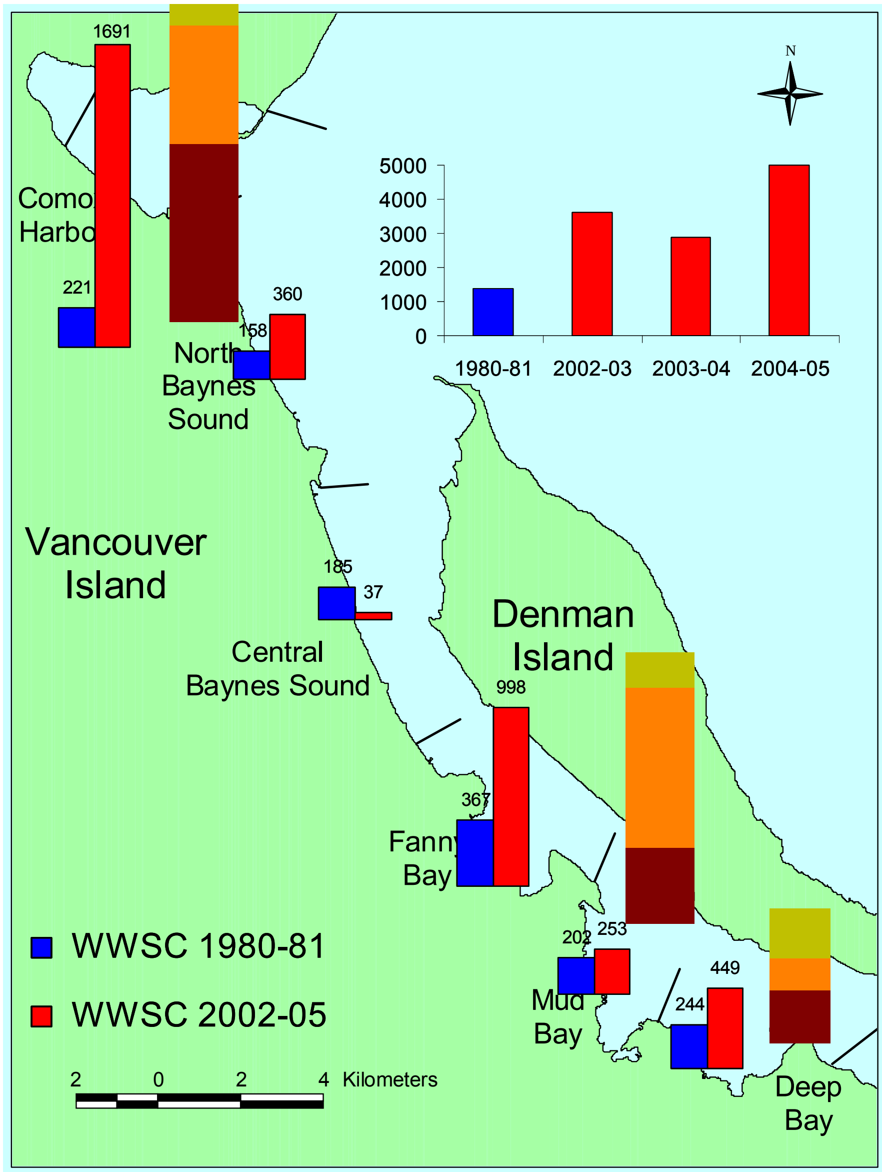


Figure 3: White-winged scoter abundance on the Vancouver Island side of Baynes Sound, WWSC = white-winged scoter. Brown bar graphs represent relative densities of bivalve species in the environment: dark brown = varnish clams, orange = manila clams, and green = native littleneck clams.

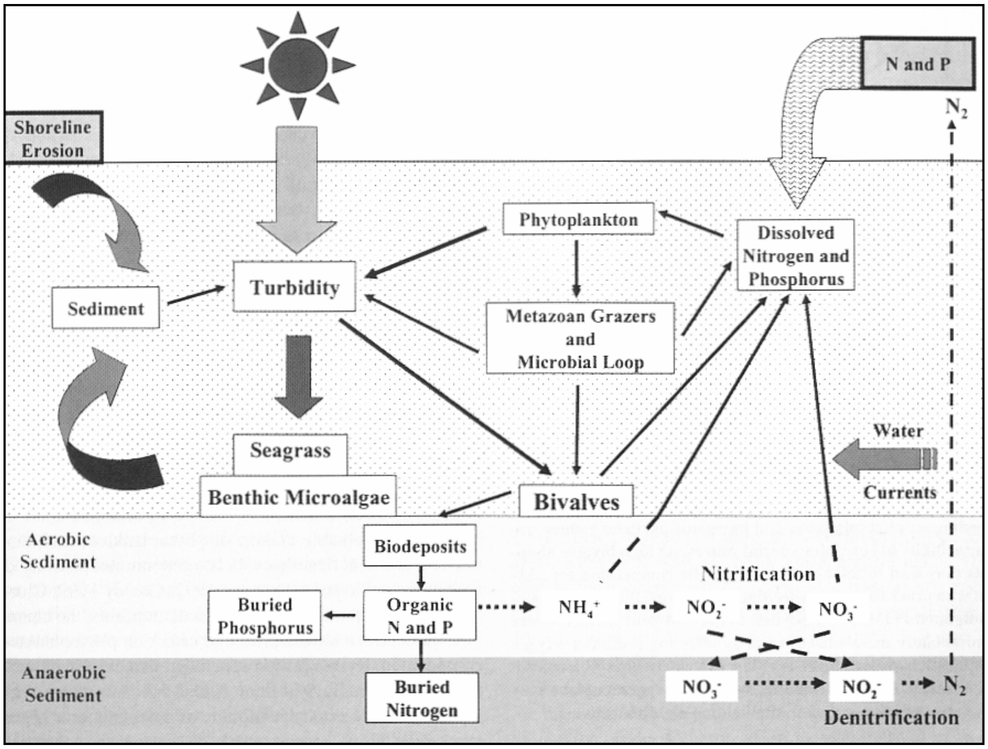


Figure 4: Conceptual diagram of the ecosystem effects of suspension-feeding bivalves. Solid lines indicate transfer of materials; dashed lines indicate diffusion of materials; dotted lines indicate microbially mediated reactions (from Newell 2004).

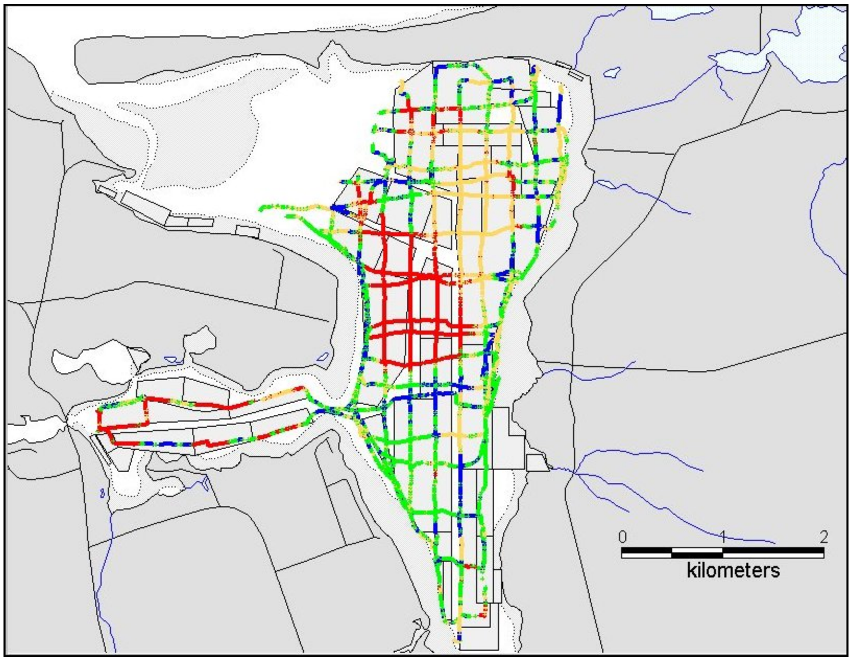


Figure 5: Acoustic seabed classification map of Tracadie Bay, PEI. This acoustic data was obtained during a survey conducted in 2002 & 2003 using the QTC View-V (50 kHz frequency with a beam width of 24º). Mussel culture leases are represented by polygons underneath the survey transect lines (Ouellette, M. unpublished data).

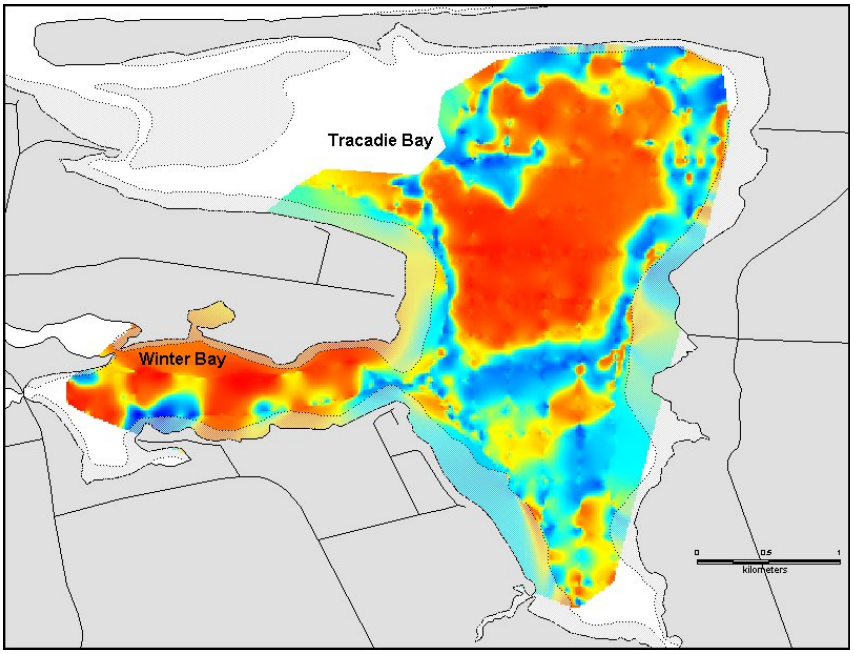


Figure 6: Interpolation map of the first principal component (Q1) of seabed acoustic features, from QTC Impact analysis, for Tracadie Bay, PEI. (Ouellette, M. unpublished data).