MARINE FINFISH AND SUSPENDED SHELLFISH AQUACULTURE: WATER QUALITY INTERACTIONS AND THE POTENTIAL FOR INTEGRATED AQUACULTURE

Thesis submitted to the University of Stirling
for the degree of Doctor of Philosophy

by

Stephen F. Cross

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DECLARATION:

This thesis has been prepared in its entirety by the Doctoral Candidate, and no part of this work has been submitted for any other degree.

Doctoral Candidate:

Stephen F. Cross

ABSTRACT

The objective of this study was to quantitatively document the culture performance and tissue quality of commercially important deepwater shellfish species (i.e., Pacific oyster, *Crassostrea gigas*; and Japanese scallop, *Patinopecten yessoensis*) cultured adjacent to marine finfish aquaculture operations, and to determine (from a production viability and seafood safety perspective) whether integrated finfish-shellfish Multi-Trophic Aquaculture (MTA), or polyculture, is a viable option for the aquaculture industry of temperate regions.

Two study sites were employed for this research, one comprising an Atlantic salmon production facility and the other a Pacific salmon farm. A 2-year assessment program for these sites detailed: (i) oceanographic and physiographic characteristics; (ii) organic waste flux, composition and dispersion; (iii) shellfish uptake and contaminant persistence; and (iv) shellfish culture performance.

Organic waste flux ranged from 17.11 g/m^2/day to 18.35 g/m^2/day at the study sites. Phosphorus, calcium, carbon, zinc, cadmium, and strontium were waste constituents that were found at elevated levels at the farm sites with significant declines in concentrations with distance downstream (maximum effect to 100-115 meters).

A mass balance estimation suggested that 85.1% of the organic material (feed) entering the cage was used for fish growth/respiration, 6.8% was lost as settlable solids to the seafloor, and the remaining 8.1% was retained in the water column and a fraction that could affect non-target species (e.g., polyculture candidates) either directly or indirectly. A similar mass balance evaluation for trace metal and chemotherepeutic constituents indicated that 12.9% of the zinc, and an estimated 98.6% of the oxytetracycline contained in feed (during treatment), was released to the water column for potential uptake by co-cultured shellfish.

The shellfish monitoring component of this study revealed that trace metal constituents of the feed did become available to the shellfish, although the quantifiable

accumulation of trace metals in these non-target species occurred only in close proximity to the cage system and only for the tested scallops (*Patinopectin yessoensis*).

Uptake rates of OTC by shellfish ranged from 0.056 – 0.100 ug/g/day with an associated clearance rate of 0.016 – 0.109 ug/g/day for the respective treatment periods. The comparison of uptake-clearance dynamics suggested a significant seasonal component to these processes. The physical and biological processes affecting contaminant uptake and clearance rates were identified as important considerations in the management of a proposed *integrated*-MTA system. A simple *Probable Effects Duration* (PED) model was developed on the basis of Uptake-Clearance-Persistence plots, illustrating the basis upon which temporal effects of water quality deterioration could be managed in such a system.

Shellfish growth was neither impeded nor enhanced as a result of being cultured directly within the influences of a salmon aquaculture facility. An organoleptic test demonstrated that shellfish palatability was not negatively impacted as a consequence of culture proximity to a finfish aquaculture facility.

Results of this research suggested that two options are available for developing MTA in coastal temperate waters, i.e., an *integrated* MTA system and/or an *adjacent* MTA system. A wide range of MTA social, technical and economic benefits were identified and discussed as a result of this research.

It was concluded that the development of a balanced MTA could add measurable environmental benefits to existing aquaculture systems, setting the stage for future production efficiencies and growth. Given a proper regulatory framework, including seafood (MTA products) and environmental quality surveillance, the potential water quality impacts on the shellfish component of a finfish-shellfish MTA (identified in this research initiative), and the associated risks over seafood safety, could be effectively managed to support this aquaculture evolution.

ACKNOWLEDGEMENTS

I would like to acknowledge and thank the Aquaculture Institute, University of Stirling, and particularly Drs. Telfer and Muir, who entrusted me with the independent task of completing this research initiative with the minimum of supervision. As a mature student I have enjoyed the learning environment offered through this relationship, and trust that a future collaborative relationship will build from this positive experience.

This research was funded jointly through the Canadian Department of Fisheries and Oceans (DFO) Aquaculture Collaborative Research and Development Program (ACRDP) and the National Research Council (NRC) Industrial Research Assistance Program (IRAP). This project could not have been completed without the support of these agencies. It is hoped that results from this research can be used to assist in shaping Canadian policies and regulations that encourages the sustainable development of aquaculture (and integrated aquaculture) in coastal Canadian waters.

Special thanks is conveyed to Marine Harvest Canada and to Heritage Salmon Limited for the use of the two salmon aquaculture sites employed in this research program. I would also like to express my sincere gratitude for the extensive operational data provided by each of these companies, and for the unconditional support of the research and of its findings.

AQUAMETRIX RESEARCH LTD. provided all equipment resources for this research (support vessels, sampling gear, current meters, laboratory space) as well as field assistants for the 2-year data acquisition component of the research. Thanks are extended to G. Gibson and N. Blasco for their technical assistance, which was provided in rain or in shine.

ALS Analytical Services completed the trace metal, antibiotic (OTC) and pesticide (emamectin benzoate) analyses. The Canadian Food Inspection Agency (CFIA) provided access to their historical shellfish tissue database that was used to assess the levels of trace metals found as a result of analyses conducted during this research program.

The British Columbia Ministry of Agriculture, Food and Fisheries (BCMAFF) conducted a single-beam acoustic survey (QT-View) of each study site in assistance of the physiographic characterization of the sites. The unprocessed data from these surveys was supplied without reservation, but rather in anticipation of receiving the results of this research for consideration under their regional, regulatory mandate.

Recognition of volunteers from the Ka:'yu:'k't'h' / Che:k:tles7et'h' coastal First Nation people for their enthusiastic participation in the organoleptic component of the project is also extended, with a special thanks to Nancy Gillette (Band Chief) for assisting in the arrangements for the test.

On a personal note, I would like to thank my wife Lisa for her continual support and encouragement of this research initiative, and to my wonderful children (Nikolas and Morgan) for sharing the *"pain of homework"* with a Dad who was *"still in school"*.

I thus dedicate this research to my family - Lisa, Nikolas and Morgan.

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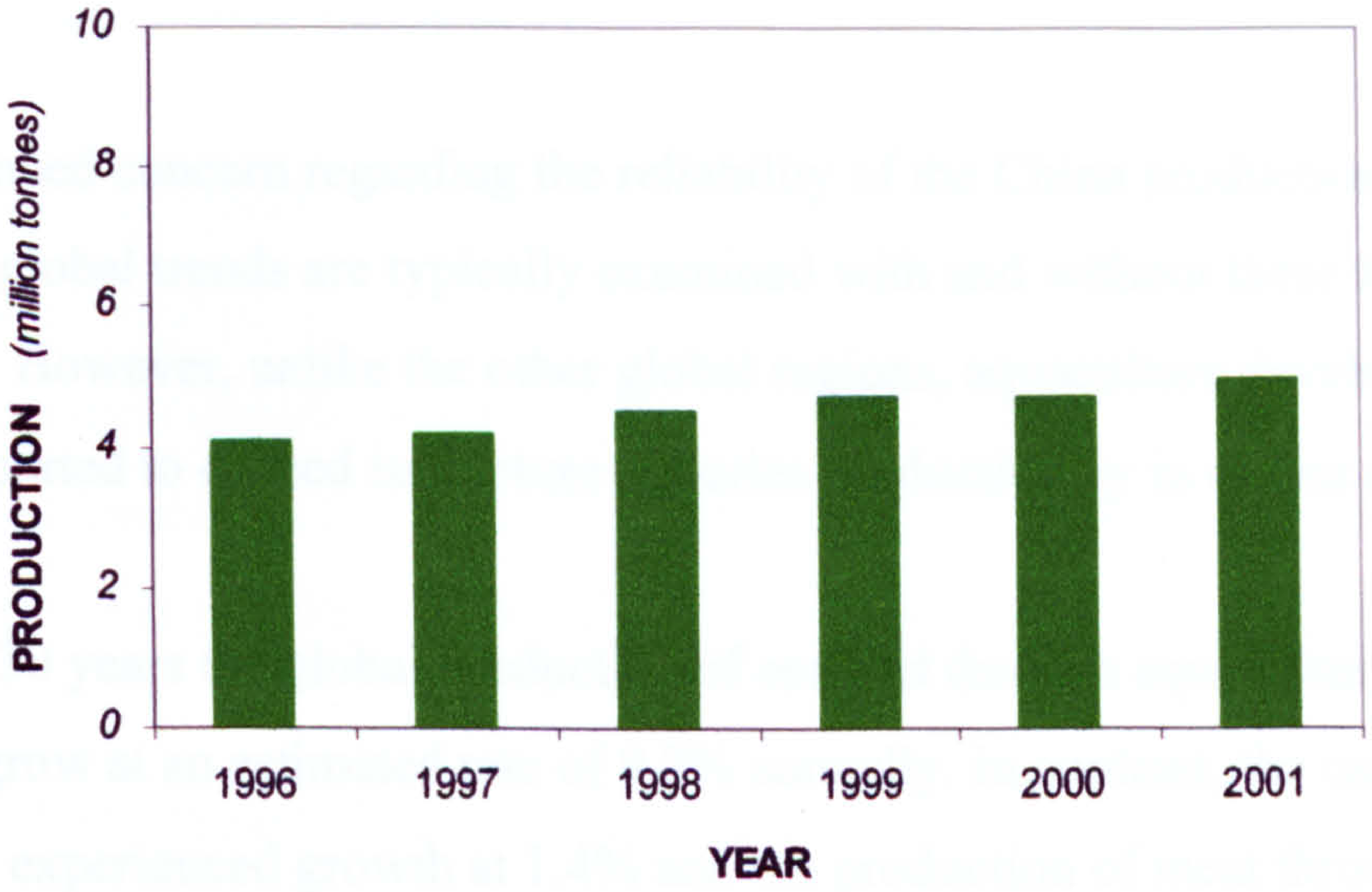
CHAPTER 1

General Introduction

1.1 Marine Aquaculture Industry

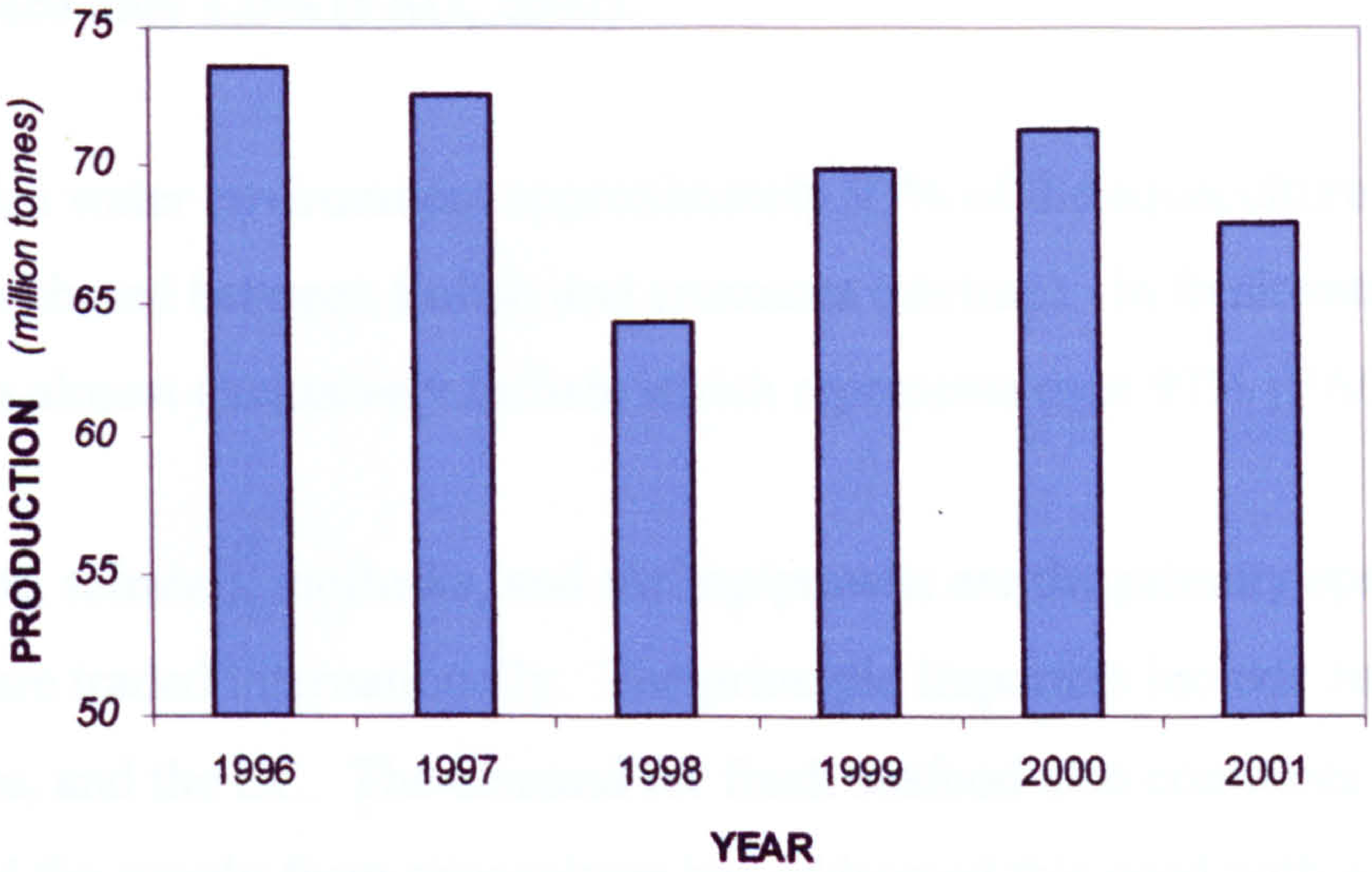
The State of the World Fisheries and Aquaculture report (FAO, 2002) provides a detailed summary of the production and market trends in capture fisheries and in global/regional aquaculture. It is clear that marine aquaculture continues to increase its seafood production share with the capture fisheries of the world, revealing an average annual production increase of 5.1%; production in 2001 exceeded 5.0 million tonnes (Figure 1).

Figure 1: Total global marine aquaculture production, 1996 through 2001. (Data from: FAO, 2002).



The marine capture fisheries remain dominant in all regions, with the exception of China, showing a globally stable level of annual production despite large differences among regions (Figure 2). In the Northwest Pacific, for example, landings have experienced a serious decline over the past 10 years, in the Northeast Pacific they fluctuate, in the West Central Pacific and Indian Oceans they have reported significant inclines over the past 30 years, and in the Northeastern Atlantic they appear very stable during the past 3 decades.

Figure 2: Total global marine capture fisheries production, 1996 through 2001. (Data from: FAO, 2002).



There is continued concern regarding the reliability of the China production statistics, and global trends are typically examined with and without these figures (FAO, 2002). However, unlike the other global regions, aquaculture development in China is reported to exceed its capture fisheries production by in excess of 27%.

Over the past 30 years the global production of seafood through aquaculture has continued to grow at an estimated rate of 9.2% annually. In contrast, the capture fisheries have experienced growth at 1.4% and the production of meat through terrestrial farming at 2.8% (FAO, 2002). In 2000 aquaculture production exceeded 45.7 million tonnes with an estimated value of US$56.5 billion.

Four major aquaculture species groups, represented by over 210 individual species, are cultured globally (FAO, 2002). These groups comprise finfish, mollusks, crustacea, and aquatic plants. Other, incidental species comprise a very small fraction of the total production and are currently viewed primarily as either locally important or experimental in nature.

The proportion of the four aquaculture groups among the aquatic habitats, freshwater, brackish water and marine waters, is portrayed in Figure 3. Each environment is dominated by different categories, with marine aquaculture producing primarily mollusks and aquatic plants, for a combined contribution of over 90%. Finfish represents 8.7% of the marine production, while crustacean species produce only 1.0% (FAO, 2002).

In the brackish water environment approximately 93% of the aquaculture production is shared between finfish and crustacea (shrimp). In freshwater the production is almost exclusively finfish, which represents over 97% (FAO, 2002).

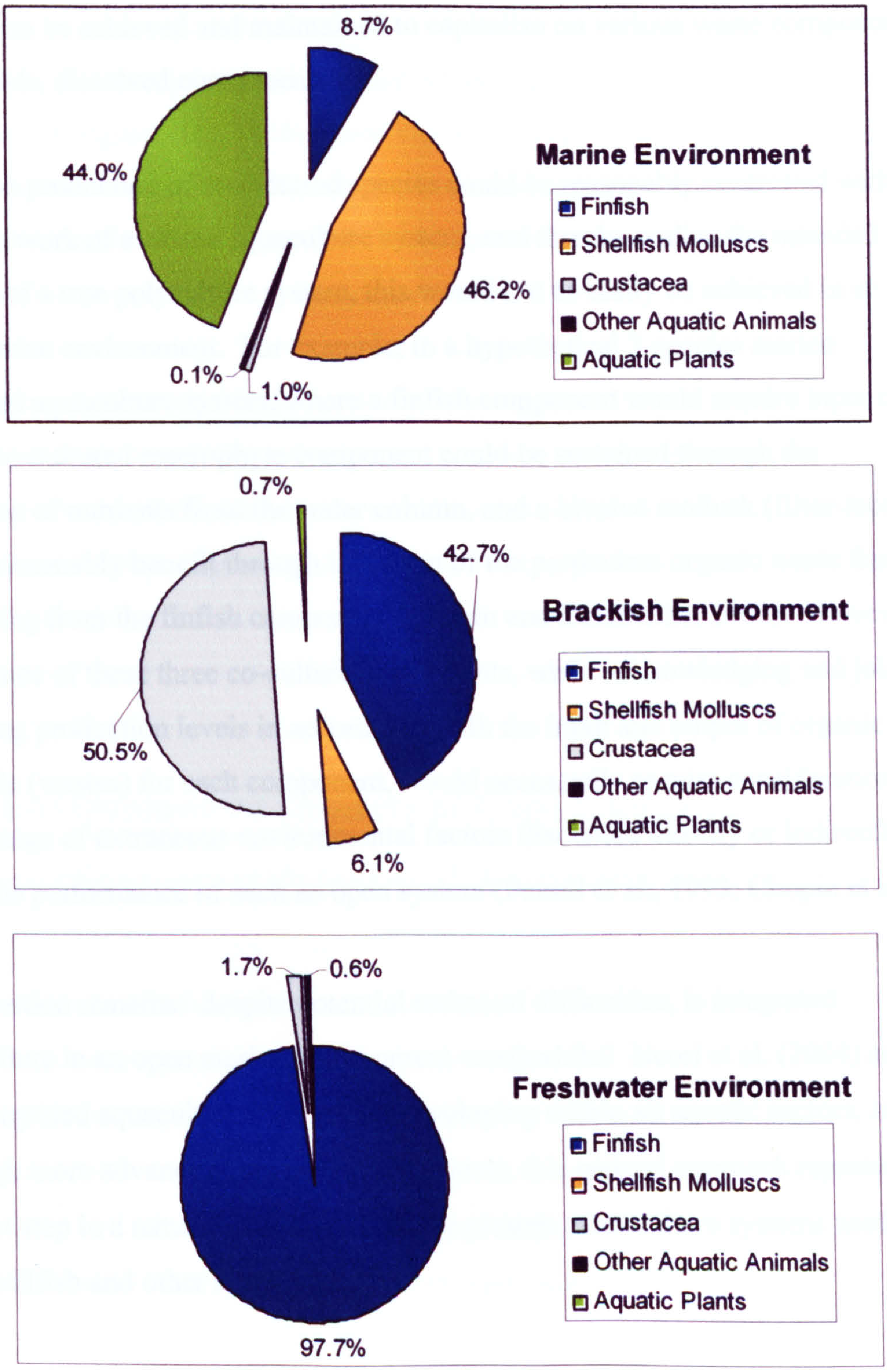
Finfish (carps, salmon), mollusks, and shrimp/prawns are the primary species groups that are traded internationally. The principle importers include Japan, the United States, and the EC. The demand for fresh seafood also continues to increase, and the supply from aquaculture has addressed this need with a global per capita supply increase from 0.6 kg in 1970 to 2.3 kg in 2000 (FAO, 2002).

As concluded by FAO (2000, 2002) marine aquaculture, over the past 30 years, has (and continues) to expand, diversify and intensify globally. With technological and husbandry innovation this sector is providing ongoing potential for meeting increasing food demand, and as a result realizing economic benefits, increased trade, improvement in standard of living, and new opportunities for rural growth.

1.1.1 Temperate Marine Aquaculture – is there a Need for Integrated Production Systems ?

The integration of multiple species into a common aquatic food production system, i.e., polyculture, has been employed in Asia (and particularly China) for many centuries. The practise of polyculture has resulted in the design and operation of systems that have demonstrated increased levels of production over that of monoculture systems (Behrends, et.al., 1985; Pavel, et.al., 1985; Perry and Tarver, 1987), with the majority of polyculture systems developed for freshwater, brackish water, and seawater ponds.

Figure 3: Major categories of species produced in aquaculture systems globally. Proportions shown for marine, brackish and freshwater environments. Data from: FAO, 2002.



Integrated culture, whether terrestrial or aquatic, has been defined as a farming practice that uses the output of one system component, which would have otherwise been wasted, to support another component and thereby increase efficiency in the overall system performance (Neori et al., 2004). A balance between integrated species can be achieved and maintained to capitalize on various waste components (e.g., solids, dissolved components).

While the production of co-cultured species could be reasonably controlled within the framework of a *closed* aquaculture system, and thereby realize the intended benefits of a true polyculture system, this would not as easily be achieved in an *open* marine environment. For example, in a hypothetical 3-species marine integrated aquaculture system, where a finfish component would require input of feed, a co-cultured macrophyte component could be sustained through the extraction of nutrients from the water column, and a bivalve mollusk (filter-feeder) could presumably benefit through ingestion of the particulate organic waste fraction originating from the finfish component (Chopin and Bastarache, 2002). However, the balance of these three co-culture components, while acknowledging and jointly managing production levels in accordance with the input and output of organic materials (wastes) for each component, would necessarily require consideration of a broad range of extraneous environmental factors that could directly or indirectly affect the performance of such an open system (Petrell et al., 1993; Chopin et al., 2001).

The question remains: despite potential technical difficulties, is integrated aquaculture in an open marine environment worthwhile? Neori et al. (2004) argued that integrated aquaculture is currently developing within all aquatic sectors, and although more advanced for freshwater systems, this general approach represents the next step in a natural evolution from the present monoculture systems used for fish, shellfish and other species.

In terms of environmental benefits, proper design of an integrated aquaculture facility (whether a closed or open marine system) will result in a natural biofiltration system for organic wastes generated and discharged from a finfish component. As described previously, incorporation of a macrophyte would extract dissolved nutrients directly from the water column, filter-feeding component would utilize the fine particulate fraction, with the combination leaving a clean water column that could be re-circulated through the finfish system component with tidal exchange and thereby maintain (or enhance) growing water conditions for this system component..

Economically, development of integrated aquaculture allows increased seafood production in regions such as New Brunswick (eastern Canada) that are considered spatially limited (in terms of industry expansion potential). In other situations it may provide opportunity for product diversification and greater efficiencies in the use of corporate capital.

Socially, the diverse opportunities offered by integrated aquaculture (multiple species) provide potential benefits directly to small coastal communities that often suffer in the wake of fisheries or other resource extraction declines. The sustainable nature of aquaculture, in general, offers considerably greater long-term stability to such communities.

1.2 Environmental Impacts of Cage Culture and the Implications to Integrated Aquaculture

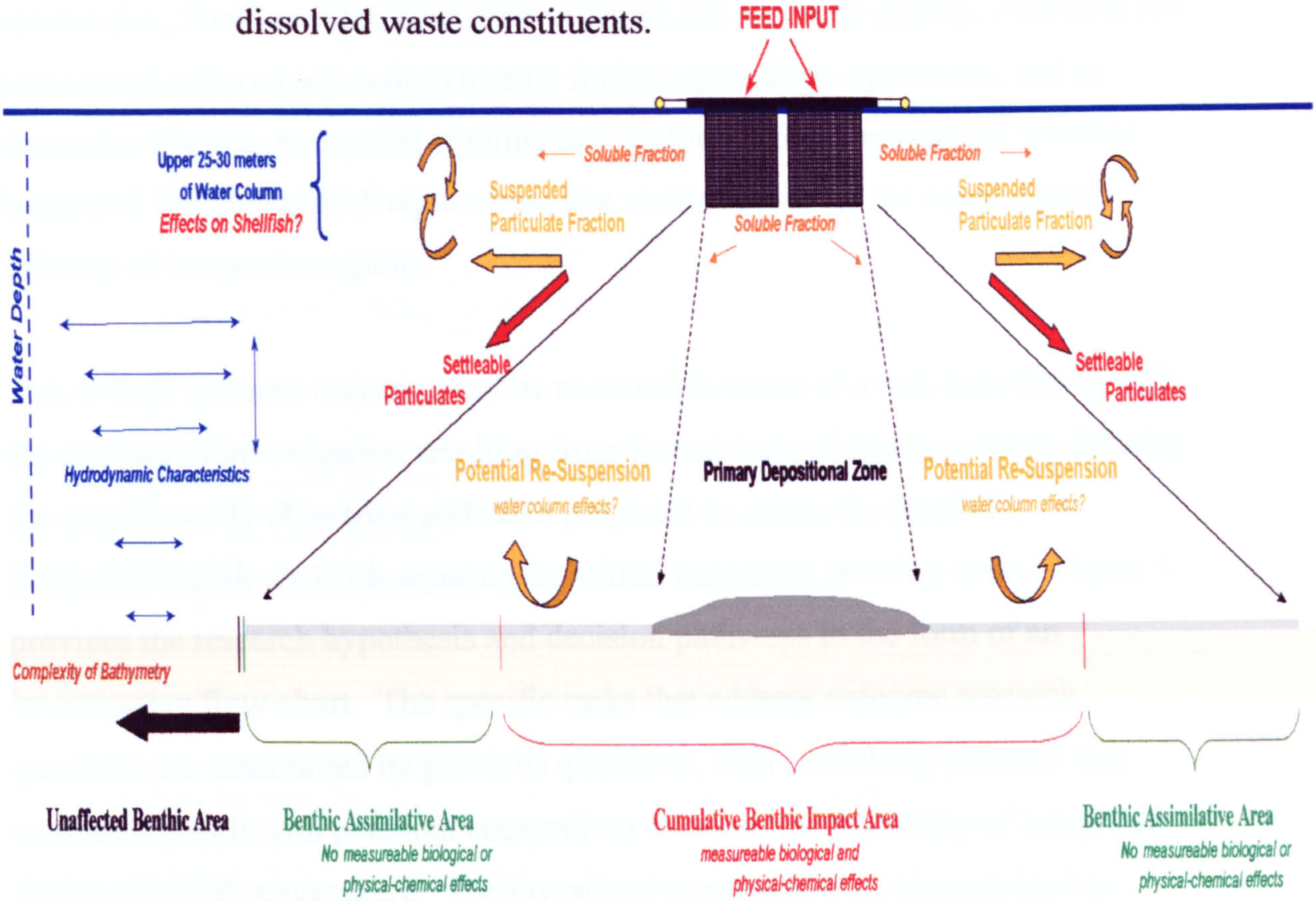
The production of finfish in temperate latitudes, which is dominated by salmonids, has used open netcage systems in suitable coastal marine areas for approximately 3 decades. The environmental impacts associated with these systems have been well documented, with reviews of these effects published intermittently over the past decade and a half (Rosenthal, 1988; Pillay, 1992; Rosenthal *et.al.*, 1995; Levings, 1994; Morrisey et al., 2000; BCEAO, 1997; Hargrave *et.al.*, 2003).

The input and subsequent conversion of feed represents the process by which organic wastes are generated from a marine finfish aquaculture facility. Organic wastes are lost to the environment as faeces, urine and a small fraction as unused feed (Gowan and Bradbury, 1987). These components represent the primary source of wastes from these farm operations, and the magnitude of potential loading to the environment is a function of farm size (level of production) and farm operational protocols (e.g., species of fish, culture density, feeding rates/practices, etc.).

Figure 4 provides a diagrammatic representation of the processes associated with the environmental fate of organic waste loss from a finfish farm. The dispersion, dilution and assimilative processes that will determine the persistence and magnitude of impacts associated with these wastes will be influenced by the oceanographic and physiographic characteristics of a farm site. The local bathymetry, topography and tidal processes at a farm site will define the accumulation "foot-print" for organic waste discharges, the change in sediment chemistry (Brooks, 2001; Hargrave et al., 1997; Wildish et al., 1999), and hence the biological response in the benthic community structure and function (Sutherland et al., 2001). These processes may also contribute to the re-suspension of these wastes during subsequent tidal exchanges (Chromey, 2002), providing a potential dynamic in the benthic zone of impact as well as contributing to potential physical-chemical and/or biological effects within the overlying water column (Chou et al., 2002; Whyte et al., 1999).

Although a wealth of research has focused on the fate and effects of the organic waste solids lost from open net-cage aquaculture systems, comparatively little attention has been dedicated to the environmental effects of the dissolved, or suspended (fine particulate) fraction that may result in localized water column effects or potentially in far-field impacts on ecosystem components. Nutrient release will most likely dilute and be assimilated very quickly within the near-field regions of these farm facilities, with little if any measurable (negative) effect on adjacent biological resources. However, the spatial and temporal biological effects of trace contaminants that may be associated with this waste component are less understood (Burridge et al. 1999) and may represent the greatest water quality risk to the development of integrated aquaculture.

Figure 4: Diagrammatic representation of waste material dilution and dispersion processes associated with a marine finfish aquaculture facility. Side view of system showing feed input, and the fate and potential effects of solid wastes (settleable component), suspended particulates, and dissolved waste constituents.



In particular supplements within the initial feed formulation, including micronutrients (trace metals), antibiotics, and chemotherapeutic compounds, will all be released to the marine environment via the organic waste pathways illustrated in Figure 4. Although typically in very low concentrations, and released at differing rates (continuous as with nutritional supplements or periodic as in therapeutic applications), the localized persistence and bioaccumulation potential of these constituents may be an important seafood safety considerations in the design (species selection, infrastructure configuration) and management of a proposed integrated aquaculture system. An understanding of these water quality interactions is essential to such a development, and is the focus of the current research initiative.

1.4 Research Objectives

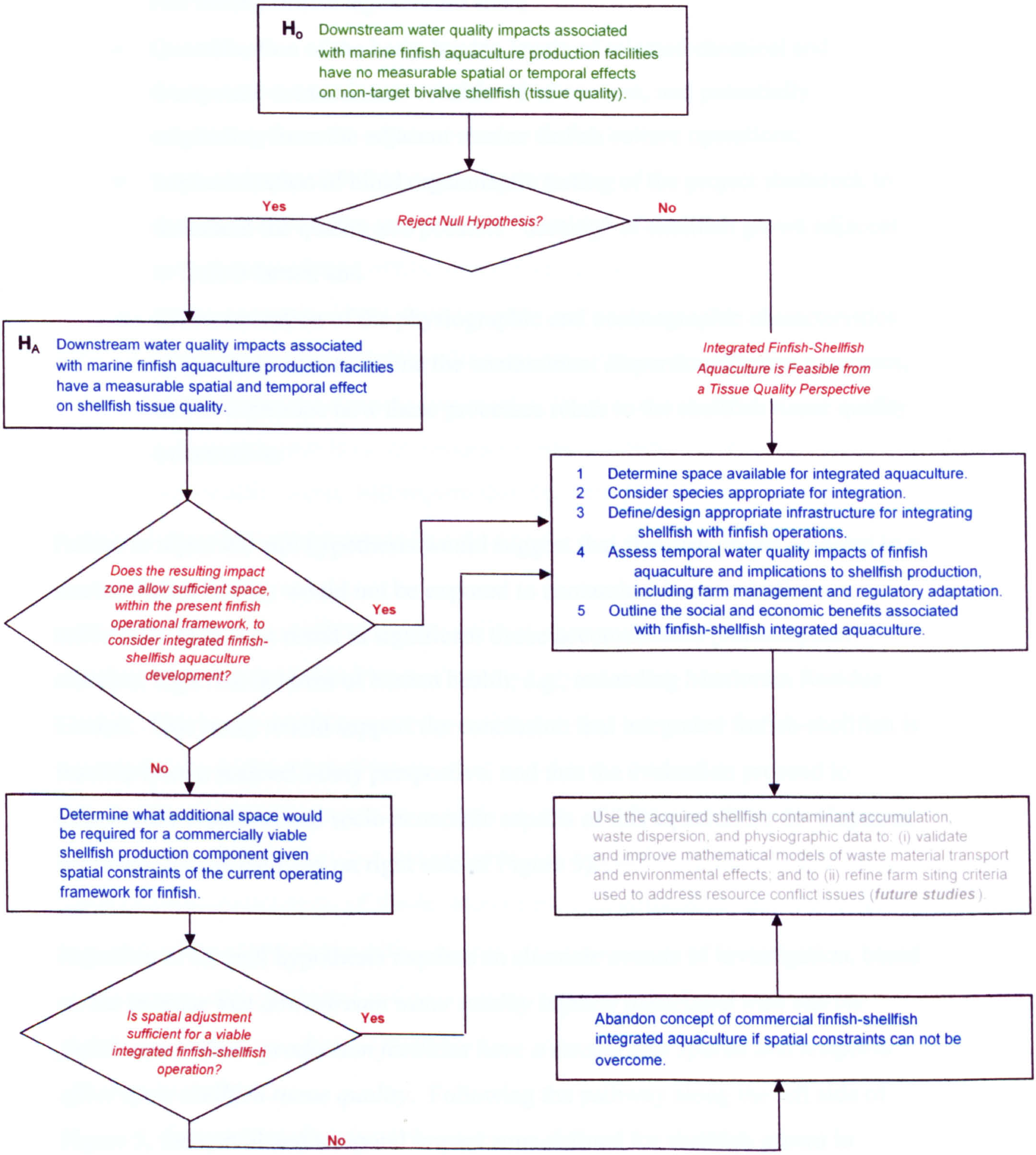
The primary objective of this study is to quantitatively document the culture performance and tissue quality of commercially important deepwater shellfish species (i.e., Pacific oyster, *Crassostrea gigas*; and Japanese scallop, *Patinopecten yessoensis*) cultured adjacent to marine finfish aquaculture operations, and to determine (from a production viability and seafood safety perspective) whether integrated finfish-shellfish aquaculture is a viable option for the aquaculture industry of temperate regions.

This overall research objective can be re-stated in terms of a null hypothesis, with the avenues of investigation resulting from formal tests of this hypothesis defining the specific study objectives and tasks proposed to assess the feasibility of integrated finfish-shellfish aquaculture within temperate growing areas. Figure 5 provides the research hypothesis and decision pathways in the form of an investigative flow-chart. The specific tasks that address outcome research questions are determined by previous questions, with a resulting pathway that continues (albeit with potential constraints) to assess the feasibility of integrated finfish-shellfish aquaculture. The investigative pathways are summarized as follows, beginning with the research null hypothesis.

H_0: *Downstream water quality impacts associated with marine finfish aquaculture production facilities have no measurable spatial or temporal effects on shellfish tissue quality.* This hypothesis, in assessing the potential of integrated finfish-shellfish aquaculture, focuses on the environmental (water quality) conditions in and around a finfish facility that could have negative effects on the quality of shellfish grown in close proximity to such a facility. It is assumed, as discussed previously, that given the bioaccumulation potential of bivalve mollusks that even relatively low levels of waterborne contaminants could be biomagnified to levels within their tissues, and that these tissue burdens could represent a significant human health risk if such shellfish were harvested and processed as a seafood product.

Figure 5

Research Hypotheses, Objectives and Implementation Pathways



Specific research objectives/tasks designed to test this hypothesis included:

- Quantification of the growth rates and survival of the commercially gown deepwater culture species *Crassostrea gigas* (Pacific oyster) and *Patinopecten yessoensis* (Japanese scallop) deployed downstream from two marine finfish aquaculture sites;
- Quantification of the body-burden levels of selected chemical and therapeutic contaminants considered of concern, and potentially originating from the adjacent marine finfish culture operations;
- Implementation of blind organoleptic testing of the project shellstock to document the quality and possible "tainting" of shellfish grown adjacent to finfish farms; and
- Characterization of the physiographic and oceanographic characteristics of the study sites to define the contaminant dispersion/dilution processes, and to determine how these processes relate to the shellfish tissue quality information.

Failure to reject the null hypothesis would suggest that shellfish grown adjacent to a marine finfish facility would not be exposed to contaminant concentrations sufficiently high as to result in significant tissue accumulations (consumption criterion: high risk in terms of human health; e.g., exceeding Maximum Residue Limits). This result would support the conclusion that integrated finfish-shellfish is feasible from a seafood safety perspective, and that the evaluation proceed to examine the technical and socio-economic aspects of this aquaculture development approach (follow pathway on right side of Figure 5).

Rejection of the null hypothesis requires an alternate avenue of investigation, based on the premise that *downstream water quality impacts associated with marine finfish aquaculture production facilities have a measurable spatial and temporal effect upon shellfish tissue quality*. Following the pathway along the left side of Figure 5, the spatial and temporal impact zone defined for shellfish grown in proximity of a finfish farm would be delimited from data acquired in testing the null hypothesis. Defining and removing the zone of shellfish impact from the total free space available within the finfish operational polygon (e.g., foreshore lease, tenure, concession) will provide a gross estimate of the area available for developing the shellfish component of an integrated aquaculture plan. Avoidance of the potential water quality effect could thereby be affected through recognition of a water quality exclusion zone, and consideration of an ***adjacent*** rather than a truly ***integrated*** finfish-shellfish aquaculture opportunity.

The physical space available for a shellfish component of a co-culture aquaculture operation, whether *integrated* or *adjacent*, must be sufficiently large as to allow for a commercially viable shellfish development, and determination of this workable area becomes an important investigative avenue of this research initiative. If sufficient space is unavailable within the existing operational area, then a regulatory mechanism may be required to acquire the space necessary to meet economic criteria for considering this integrated development; examination of the optimum space requirements is considered at this point in the evaluation process. However, failure to acquire a critical level of operational space, either as a result of practical or regulatory considerations, will require that the concept of an integrated finfish-shellfish aquaculture development be abandoned.

Rejection of the initial null hypothesis will not necessarily result in the abandonment of an integrated aquaculture plan, but more likely result in the delimitation of a predictable impact zone that could be regarded as a shellfish aquaculture exclusion area within an integrated finfish-shellfish operational framework. This spatial constraint would not preclude the development of an integrated aquaculture plan, but rather would assist in defining the site-specific infrastructural relationships of the development. This alternative pathway (Figure 5) would culminate, with the initial pathway (acceptance of the null hypothesis), in an evaluation of the technical and socio-economic benefits of an integrated aquaculture approach. Specific assessment objectives, at this point of the research program, include:

- Determining the physical space available for introducing a shellfish aquaculture component to an existing finfish operation, as well as an optimum physical configuration for a new integrated finfish-shellfish aquaculture facility;

- Consideration of the appropriate species for integration given the inherent differences in biophysical capability criteria of various shellfish and finfish taxa;
- Defining the appropriate (compatible) infrastructure for integrating shellfish and finfish operations;
- Assessing the temporal water quality impacts of finfish aquaculture facilities and discussing the implications of such water quality risks with shellfish production (e.g., regulatory adaptation, perception issues); and
- Delimiting the social and economic benefits associated with integrated finfish-shellfish aquaculture operations.

The empirical data collected over the course of this research will, in addition to objectively assessing the feasibility of integrated finfish-shellfish aquaculture in temperate regions, provide an information base that can be used to further understand and manage the spatial and temporal environmental effects of marine finfish aquaculture. Data acquired on the waterborne contaminant impacts to shellfish will prove valuable in improving farm siting criteria (resource conflict avoidance), and the waste material dispersion, physiographic and oceanographic data will help to continually improve (validate) working mathematical models that proclaim to accurately predict contaminant dispersion, dilution, assimilation, and overall environmental risk. These objectives, while considered important ancillary uses for these data, are left for future evaluations and scientific publication.

The following chapters present the experimental design, analytical and field methodology, data analyses, interpretation and conclusions associated with the systematic evaluation of the feasibility of integrated finfish-shellfish aquaculture.

CHAPTER 2

Research Design and Study Sites

2.1 Research Design

In testing the hypothesis that there are no downstream contaminant impacts on shellfish resources, and thus there are no tissue quality impacts of concern in supporting a move to integrated finfish-shellfish aquaculture, the design of this research project considered the factors affecting potential contaminant loading, as well as the physical processes that determine contaminant dispersion and dilution.

2.1.1 Contaminant Loading and Bioavailability

As discussed in Chapter 1, the composition and potential loading of contaminants originating from a finfish production facility will depend primarily upon the size of the farm and the operational procedures used at each particular facility that might result in a loss of contaminants to the environment. The categories of contaminants potentially available as environmental impact agents within the water column, either as (or associated with) suspended particulates or in the form of dissolved constituents, comprise the following:

- *Trace Metals*, originating as: (i) antifouling leachates from system netting; (ii) constituents (micronutrients) of fish feed released in wastage and/or in feces; (iii) leachates of farm structures, e.g., galvanized system components; (iv) incidental losses of equipment fluids (oils, grease, fuels, paints, cleaning agents, etc.).
- *Suspended Organic Particulates*, originating from waste feed and/or from fish fecal material.
- *Hydrocarbons*, associated with leakage from equipment using internal combustion engines (boats, generators, pumps, winches, etc.).
- *Chemotherepeutant* residues (e.g., antibiotics, sea lice treatment compounds) released with wasted feed, fish feces.
- *Disinfectants*, loss to the environment during direct application and discharge (e.g., biosecurity measures).
- *Bacteria and Viruses*, associated with: (i) intensive finfish husbandry and loss to the environment during a disease outbreak event; and with

(ii) human wastes, generated from site staff origin and lost via direct sewage discharge.

- *Complex Organic Compounds*, such as dioxins, PCB's, that may be associated (at low concentrations) with the fish meal component of feed and subsequently lost to the environment with fecal material and/or wasted feed pellets.

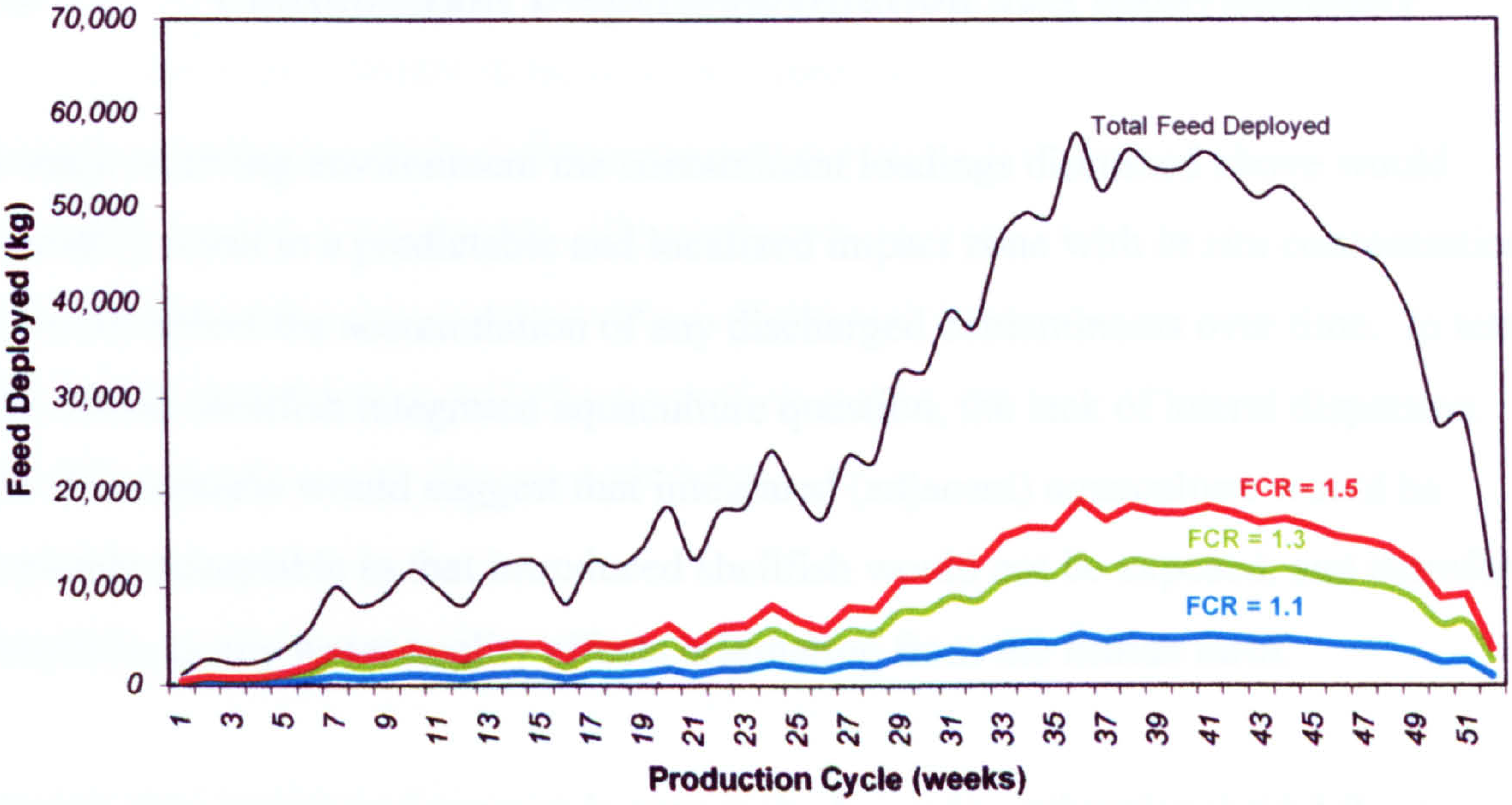
The environmental loading and subsequent fate and effects of these farm-derived contaminants have a distinct temporal component, related to the production cycle of the farm site, but the actual magnitude of these contaminant loadings will also be dependent upon the operational efficiencies and due-diligence in site-specific husbandry practices. Many of the above contaminants (e.g., fuel residues, disinfectants, trace metals from nets) may never be lost to the environment, or the risk of discharge may actually be extremely low, given application of best husbandry practices at the farm. For example, fuel containment may prevent or reduce risk of spills, net maintenance (cleaning) may eliminate the need for using metal-based antifoulants, use of disinfectants may be conducted in closed containment systems, and disease outbreaks may be infrequent given rigorous fish health surveillance (resulting in minimal required use of chemotherapeutic compounds and reduced bacterial or viral loads at the site).

The use of open netcage systems will, however, always result in the loss of fecal material to the environment, with the composition of the initial feeds defining the contaminant constituents of the resulting feces which will, in turn, determine the environmental risk of these agents. The loading of this contaminant component of the farm operation will also, to some degree, rely upon husbandry. The use of optimally formulated feeds through the production cycle will ensure maximum digestibility, adequate pellet sizes for size of fish, and thus the lowest possible feed conversions (FCR's) and minimal through-put (feces generation). Extension to feeding procedures themselves will ensure minimal loss of feed to the environment with use of technological innovations (e.g., camera systems) and close monitoring of feeding behavior and the environmental conditions that may disrupt such behavior.

Figure 6 illustrates the gross change in organic loading (encompassing feed waste and fecal/respiratory components) associated with change in Feed Conversion Ratio. Cumulatively, these loadings equate to 128, 324, and 468 metric tonnes of waste discharged over a period of one year, providing an indication of the potential variability in loading across farm operations in terms of organic waste flux and of the associated constituents which may become bioavailable to water column organisms such as cultured shellfish.

Figure 6

Estimate of the proportion of organic material lost to the environment (faeces, respiration) in relation to total feed deployed. Example of the difference associated with various Feed Conversion Ratios (FCR's)



The loss of organic waste is considered a key pathway for potential water quality impacts, and thus an important factor in the design of this research program. Contaminants in 5 of the 7 categories listed above are potentially available to the environment as a result of feeding (pellet loss) and the digestive process (fecal material discharge) of the standing fish biomass at the farm site. These contaminants include the: (i) trace metals, or micronutrients contained in the formulated feed; (ii) organic particulates, as feed dust or the suspended fecal fraction; (iii) chemotherepeutant residues, following a required treatment; (iv) bacteria or viruses; and the (v) complex organic constituents associated with feed quality (fish meal origin).

The other contaminant sources (e.g., site sewage discharge, net antifoulant leaching, disinfectant release, fuel spillage) are considered secondary to that of the organic waste component due primarily to the relative difference in the loading magnitude of these components. These secondary contaminant sources comprise comparatively slow release, discontinuous in nature, and in some cases only as a result of a catastrophic operational failure (e.g., fuel spill). Although not considered insignificant in nature, the design of the research program focused on the continuous waste discharge aspect of the farm operation (fecal material losses) assuming that the release of these other contaminants would follow a similar dispersion pathway to that of the organic waste material, and thus also become available to adjacent shellfish within a similar spatial distribution pattern.

2.1.2 Contaminant Dispersion/Dilution and Bioavailability

In a static receiving environment the contaminant loadings discussed above would presumably result in a predictable and localized impact zone with *in situ* concentrations that would reflect the accumulation of any discharged contaminants over time. In terms of the finfish-shellfish integrated aquaculture question, the lack of lateral dispersion under this scenario would suggest that integrated (adjacent) aquaculture would be completely acceptable in that introduced shellfish would not be exposed, and therefore susceptible, to any water quality effects originating from the finfish farm.

However, the marine environment is extremely dynamic, and regional tidal fluxes are typically expressed in a wide variety of complex patterns on a localized scale (Thomson, 1981). These site-specific processes will be determined by a combination of tidal exchange, topographic features and bathymetric characteristics. For example, constrictions of land (narrow passages) will accentuate tidal flows and result in the development of tidal jets that will vary in vertical influence depending upon the depth of the passage. In addition, the nature of the seafloor itself (e.g., boulder, bedrock, sloped, flat) will determine how water flows across this surface, which may result in localized entrainment of materials and re-suspension upwards and into the water column. These types of processes will contribute to water column mixing, both vertically and horizontally, and thus define the extent to which waterborne materials will be dispersed and diluted.

The physiographic and oceanographic characteristics of the area at and immediately surrounding a marine finfish aquaculture site are considered critically important in determining the dispersion and dilution of contaminants originating from a farm facility. These physical processes, in concert with the contaminant loadings, will determine the *in situ* concentrations of the various contaminants and the subsequent bioavailability and biomagnification potential to non-target species (shellfish for the purpose of this research).

The magnitude of tidal flows through a farm site will have a direct influence on dispersion and dilution, and the speed-direction frequency of tidal flows over a lunar cycle will determine whether there is predominate "off-set" in dispersion over time. Such patterns will also aid in defining the spatial impact zone within which shellfish aquaculture may be inappropriate. Physical oceanographic processes are inherently complex, and relating these processes to contaminant dispersion, dilution and environmental assimilation will also require an assessment of vertical as well as horizontal hydrodynamic changes with time.

2.1.3 Research Parameters

Delimiting the spatial and temporal water quality impacts of finfish production facilities, and the potential effects of these conditions on adjacent shellfish resources, was accomplished using a direct measurement of shellfish tissue, focusing on the accumulation of a variety of contaminants representative of materials released from adjacent finfish farms. Tissue levels of the following parameters were considered in this assessment:

- Trace metals (scan of 25 elements; Table 1)
- Oxytetracyline and tetracycline residues
- Emamectin benzoate

Table 1

Assessment parameters and their associated analytical detection limits

PARAMETER	***Detection Limit (ug/g)***	**PARAMETER**	***Detection Limit (ug/g)***
Aluminum T-Al	2	Antibiotics:	
Antimony T-Sb	0.01	Oxytetracycline (OTC)	0.1
Arsenic T-As	0.2	Tetracycline	0.1
Barium T-Ba	0.01	Chemotherepeutants:	
Beryllium T-Be	0.1	Ememectin benzoate	0.1
Bismuth T-Bi	0.1	Other Contaminants:	
Cadmium T-Cd	0.05	Extractable hydrocarbons	50
Calcium T-Ca	2	Bacteria:	
Chromium T-Cr	0.2	total coliforms	2 MPN
Cobalt T-Co	0.02	Vibrio species complex	
Copper T-Cu	0.2		
Lead T-Pb	0.02		
Lithium T-Li	0.1		
Magnesium T-Mg	2		
Manganese T-Mn	0.2		
Molybdenum T-Mo	0.01		
Nickel T-Ni	0.1		
Selenium T-Se	0.1		
Strontium T-Sr	0.5		
Thallium T-Tl	0.01		
Tin T-Sn	0.05		
Uranium T-U	0.002		
Vanadium T-V	0.3		
Zinc T-Zn	2		

The species of shellfish used for this assessment includes *Crassostrea gigas*, the Pacific oyster, and *Patinopecten yessoensis*, the Japanese scallop. Both species are commercially important suspended (off-bottom) aquaculture species, ensuring that results of this research could be applicable from a commercialization perspective (i.e., development of integrated finfish and suspended shellfish aquaculture employing these species).

These shellfish also represent species that fill slightly different ecological niches, with *Crassostrea gigas*, and other commercially important oyster (e.g., *Ostrea edulis*) and mussel (e.g. *Mytilus edulis*, *Mytilus galloprovinciallus*), species living sedentary spat through adult periods of their life cycles within the mid-intertidal through shallow subtidal zones. This bivalve mollusk group is typically tolerant of a wide range of environmental factors, including temperature, salinity, and exposure.

The adult form of *Patinopecten yessoensis,* in contrast, occurs naturally as a free-swimming epibenthic scallop, found typically in a deeper subtidal habitat (20 – 50 meters), and within an environment comprised of a significantly smaller range in ambient conditions than that of the shallow-water habitat occupied by *Crassostrea gigas*. The optimal water quality conditions for this species includes moderate temperatures (7-11° C), high salinity (>29 ppt), and minimal temporal variation in these factors.

The selection of these two shellfish species as assessment candidates for this research project provided a number of advantages in addressing the study objectives. The five research design benefits realized through use of these species include:

- (1) that both are commercially cultivated suspended aquaculture shellfish species, and are thus representative of species that could be developed in an integrated finfish-shellfish facility;
- (2) that they represent different shellfish from ecological niches, characterized by contrasting biophysical conditions that determine optimal growth/survival for each species, and thus provide the opportunity for documenting downstream water quality effects at different segments of the affected water column;
- (3) that the physiological attributes are sufficiently different for each species that contaminant bioaccumulation and clearance (depuration) rates, determined largely by species-specific filtration rates (Noakes, 1998), will provide a range of realistic measures of tissue burden levels of any contaminant loads available to these animals;
- (4) that, jointly, the selected species provide a variety of shellfish *products* to be evaluated from a seafood safety perspective: i.e., whole oyster tissue (shucked and cooked, eaten raw), whole scallop tissue (steamer), scallop muscle tissue (commonly used in scallop dishes), scallop muscle with rose attached (roe-on, popular in Japanese markets); and

- (5) that the above shellfish products, offered by the variety of processing methods applied to the cultured scallops, also allows for an evaluation of contaminant partitioning across a number of tissues in this animal.

2.1.4 Statistical Considerations

As stated previously, the focus of this research program was to delimit the spatial and temporal water quality impacts associated with marine netcage aquaculture facilities, and to determine if integrated finfish-shellfish aquaculture is a technically viable option to aquaculture diversification in temperate regions. The oceanographic and physiographic characteristics of a site have been identified as important factors in determining the dispersion, dilution and bioavailability of any waterborne contaminants originating from a farm operation to adjacent shellfish resources. It is assumed, from the nature of physical processes of particle dispersion and mixing, that wastes from a netcage facility will generally result in a pattern of declining *in situ* concentrations with distance from the source.

Mathematical models that describe dispersion and dilution processes are common in a variety of industry situations to assess plume characteristics and performance (Baumgartner et al. 1994; Cromey et.al., 1998), and are routinely used in the design of discharge systems (outfall specification, diffuser system configuration, etc.) that will optimize environmental assimilation of such materials and ensure that the environmental fate and effects of discharged wastes are minimized. For marine netcage aquaculture, organic waste models such as DEPOMOD/MEREMOD (Cromey, 2000) and RAM (Carswell, 2003) are used as decision-making tools for regulators, allowing estimation of waste material dispersion and benthic effects to aid in proper siting, resource conflict avoidance, etc.

The null hypothesis of this research is one of no difference in tissue quality impacts in shellfish downstream of a finfish aquaculture facility. From a statistical perspective, and assuming some predictable pattern of decline (linear, curvilinear) in particulate and/or waterborne contaminant concentration downstream of a netcage facility, this hypothesis is best tested using a regression model approach. This can, simplistically, be described as follows:

$$H_o: \; B = 0,$$

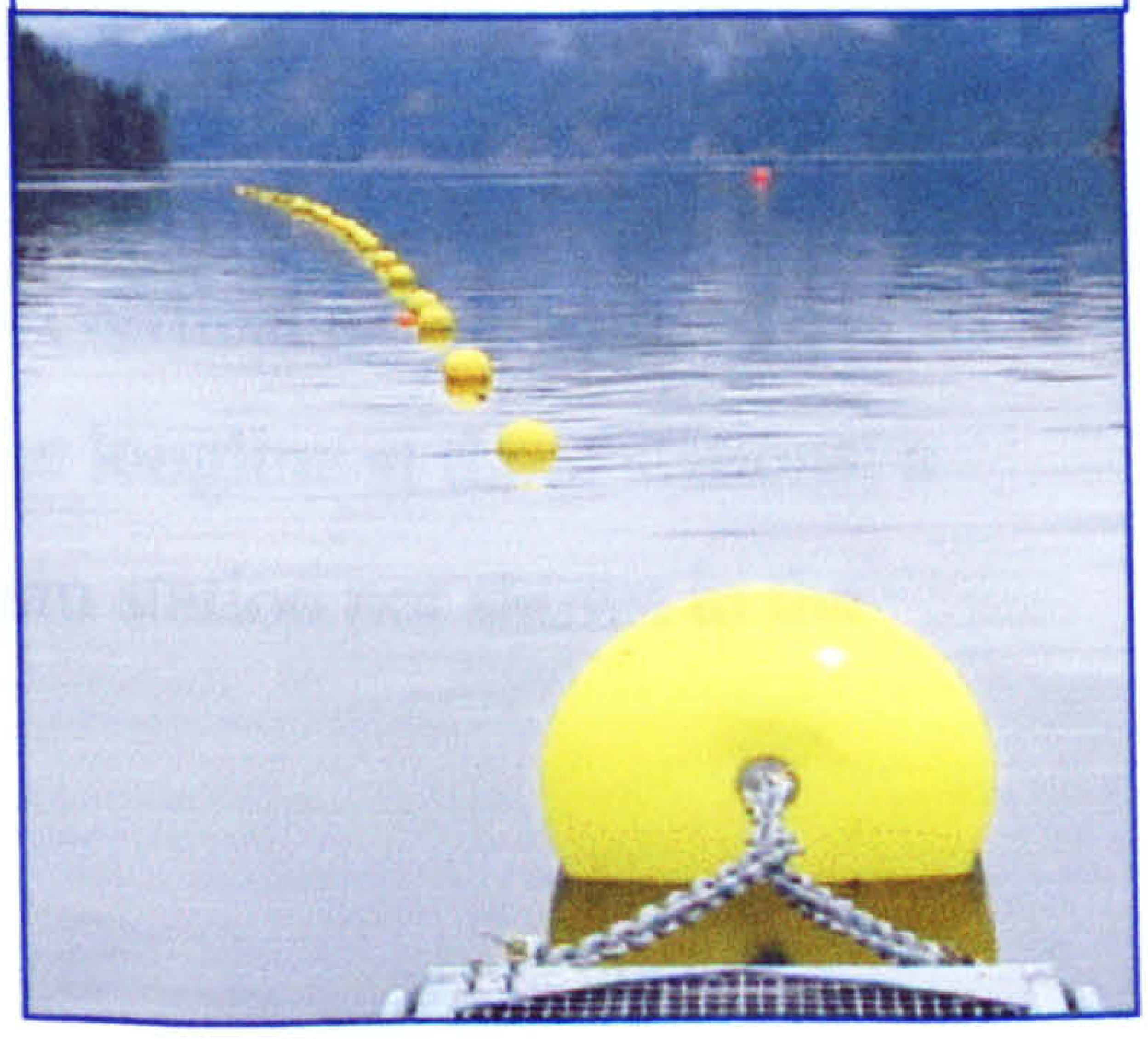
where B is the slope between tissue contaminant concentration (dependent variable) and distance from the netcage facility (independent variable). Failure to reject this hypothesis would suggest that the slope between contaminant concentrations of shellfish tissues does not show a changing pattern with distance from a finfish farm facility. This does assume, however, that sufficient distance is measured from the facility to ensure that tissue testing is not simply revealing elevated levels of contamination that are homogeneous downstream of the facility.

The implications of this simple regression model were used to design the infrastructure configuration described in the following section. However, in quantitatively assessing the spatial and temporal impacts of waterborne contaminants on downstream shellfish resources, a more complex series of statistical methods were employed. These techniques, including Principal Components Analysis (PCA), allowed a concurrent assessment of the multivariate data compiled as a result of the suite of analytical tests completed on the routinely sampled shellfish tissues. Descriptions of the specific approaches used in assessing these data are found in Chapter 4 (Contaminant Inputs and Dispersion) and Chapter 5 (Contaminant Effects on Shellfish).

2.1.5 Research Infrastructure Configuration

Sampling equipment and test bivalve mollusks were suspended in the water column from a shellfish longline (2 cm diameter polysteel rope), supported horizontally (1.0 meter below the sea surface) by regularly-spaced polyethylene floats (40 cm diameter). The longline system was aligned in the downstream direction from the finfish production facility (determined through tidal current measurements; see Section 2.2.1), attached to the

Figure 7: Shellfish research longline extending downstream from study farm site



steel cage system at one end and anchored in place with a 2-tonne cement block at the other (Figure 7 illustrates the visible features of this infrastructure). The entire floating portion of this structure was 250 meters; the distal anchor line was deployed at a scope of 3:1.

Figure 8 provides a diagrammatic representation of this infrastructure showing the plan-view and side-view configuration of sampling infrastructure in relation to the sea surface, sea bottom, and the adjacent netcage system.

Figure 8A (upper portion) illustrates the position of the sampling stations that were established along the subsurface shellfish longline. A total of ten stations were deployed along the line at each of the two study sites. Sampling stations were concentrated in the near-field region of the netcage where waste material and waterborne contaminant effects were presumed to be the greatest. These stations were established at the netcage perimeter (0-meters), and then downstream at 10, 20, 30, 50, 75, 100, 125, 175 and 225 meters from the edge of the netcage system.

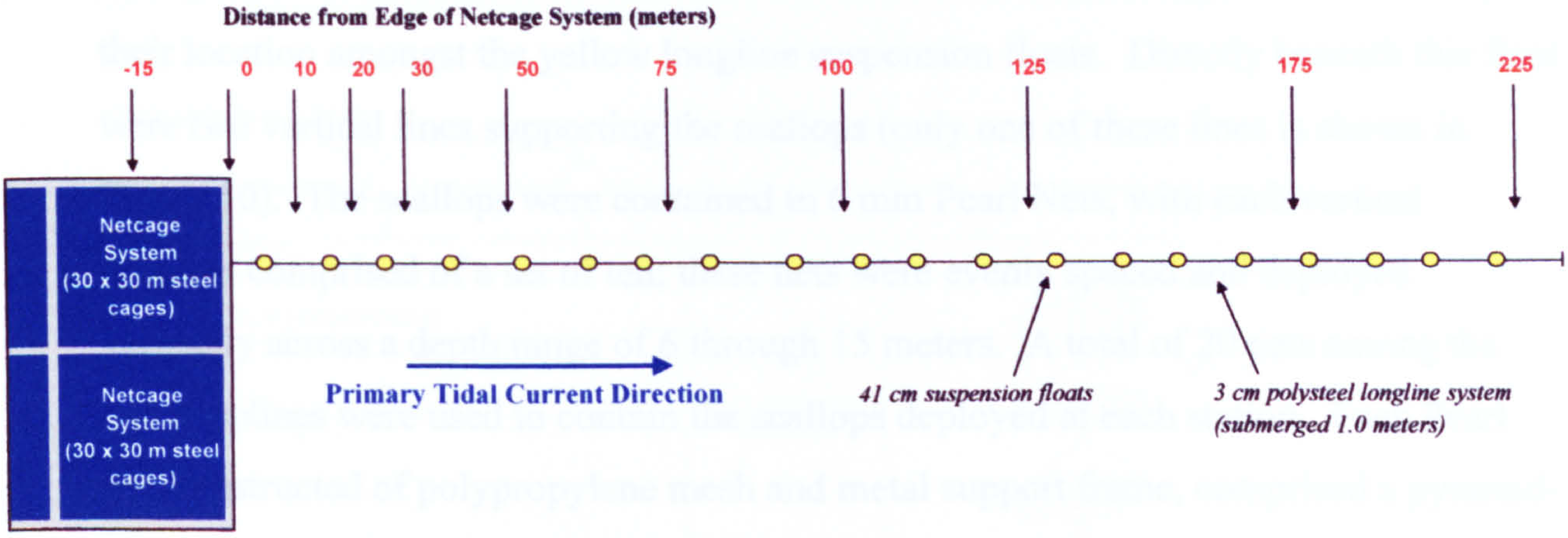
An eleventh sampling station was deployed within the nearest netcage at each of the sites. This station was considered a positive control to the study as all of its constituent sampling apparatus would be in direct contact with the materials entering the farm system, i.e., feed, fecal material (and its leachates), trace metals from surrounding treated nets, etc.

The marine netcage systems at the study sites employed in this research initiative comprised steel cages that measured 30 x 30 meters, with a depth of approximately 22 meters. Figure 8B (lower portion) provides a cross-sectional view of the steel netcage system in relation to the established shellfish longline infrastructure. The vertical area occupied by the sampling gear at each of the first three stations (in-cage, perimeter, and at 10 meters) is delimited with the green rectangular box (vertical axis elongated). This is the area in which the shellfish were suspended from the longline at these stations; a similar configuration was used for each of the downstream station not shown in the figure.

Figure 8

Research infrastructure configuration showing shellfish longline system in relation to netcage aquaculture facility. **A.** Horizontal distribution of sampling infrastructure; **B.** Vertical distribution of sampling infrastructure in relation to farm structures.

A. Plan-View Diagram of Netcage System in Relation to Experimental Longline and Sampling Stations.



B. Side-View Diagram of Netcage System in Relation to Closest Samplng Infrastructure.

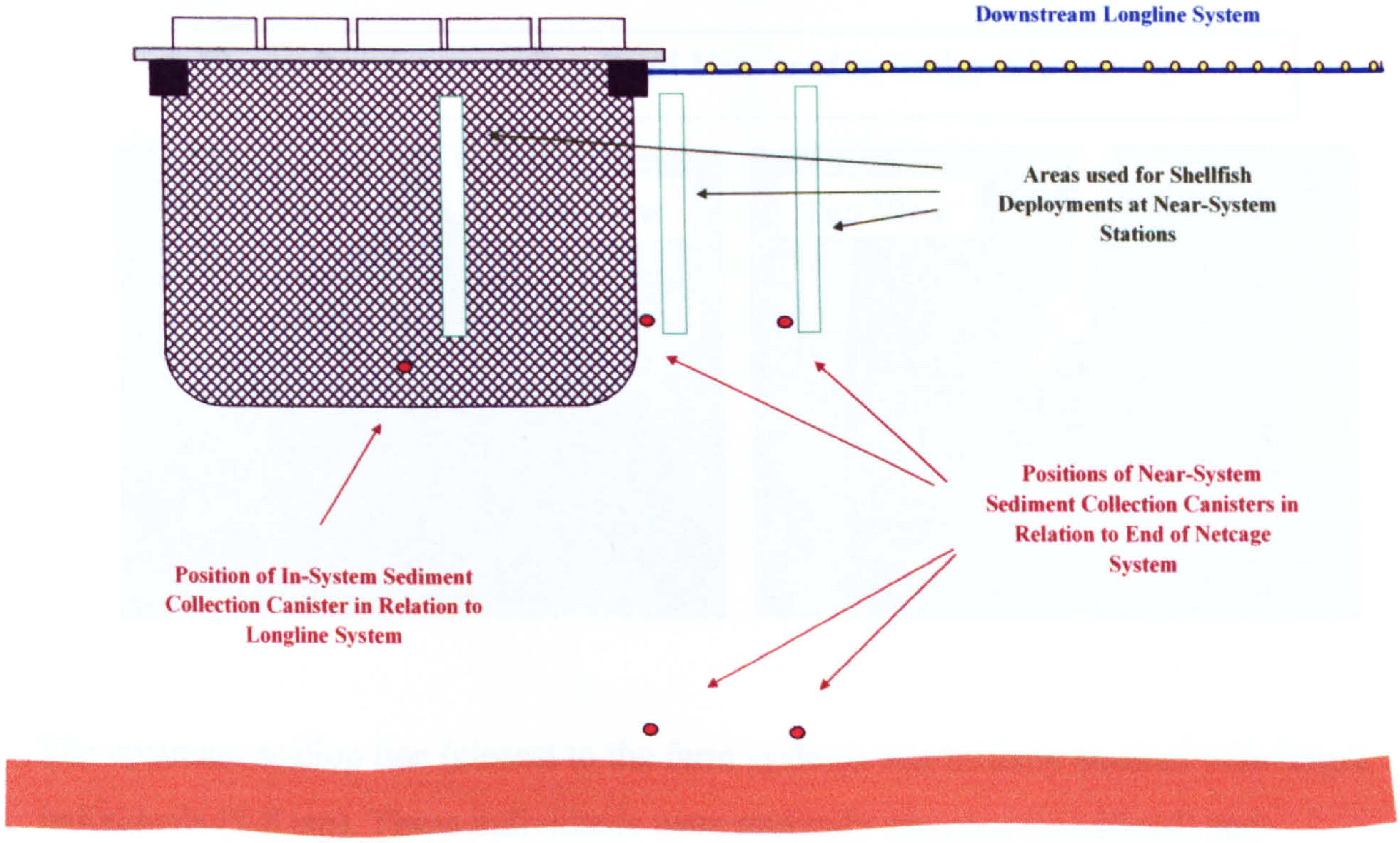
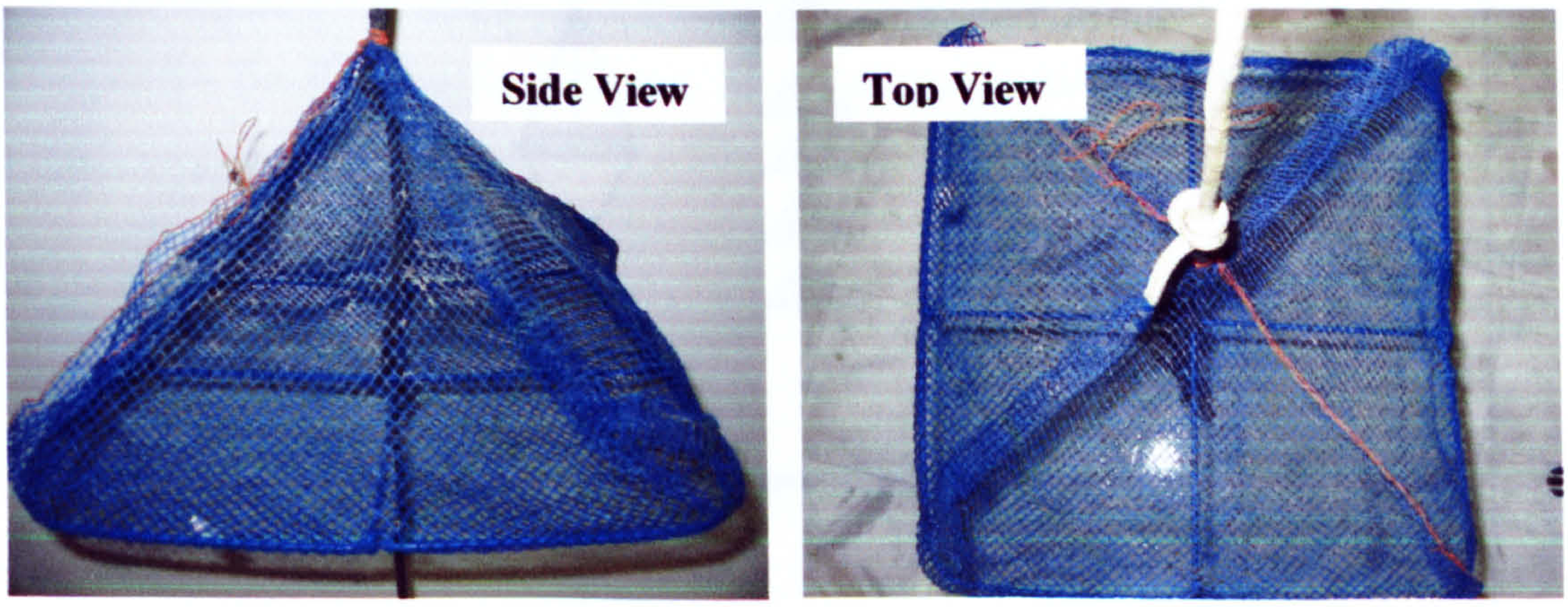


Figure 8B also shows the position of sediment collection canisters, which were deployed to measure the rate and loading of organic waste material along the dispersion pathway established for the shellfish longline. These canisters were deployed at a depth of 15 meters, close to the bottom of the shellfish deployment area, as well as 5 meters above the seafloor. A description of these canisters, and their intended use, is provided in Section 2.2.2.

Figure 10 presents a schematic representation of the sampling gear deployed at each survey station. Individual stations were marked with a small orange float to identify their location amongst the yellow longline suspension floats. Directly beneath this float were two vertical lines supporting the scallops (only one of these lines is shown in Figure 10). The scallops were contained in 6 mm Pearl Nets, with each vertical dropline comprised of a set of ten; these nets were evenly spaced and deployed vertically across a depth range of 6 through 15 meters. A total of 20 nets among the two droplines were used to contain the scallops deployed at each station. Each Pearl net, constructed of polypropylene mesh and metal support frame, comprised a pyramid-shaped containment chamber with a 40 x 40 cm base and an elevation of approximately 35 cm (see Figure 9).

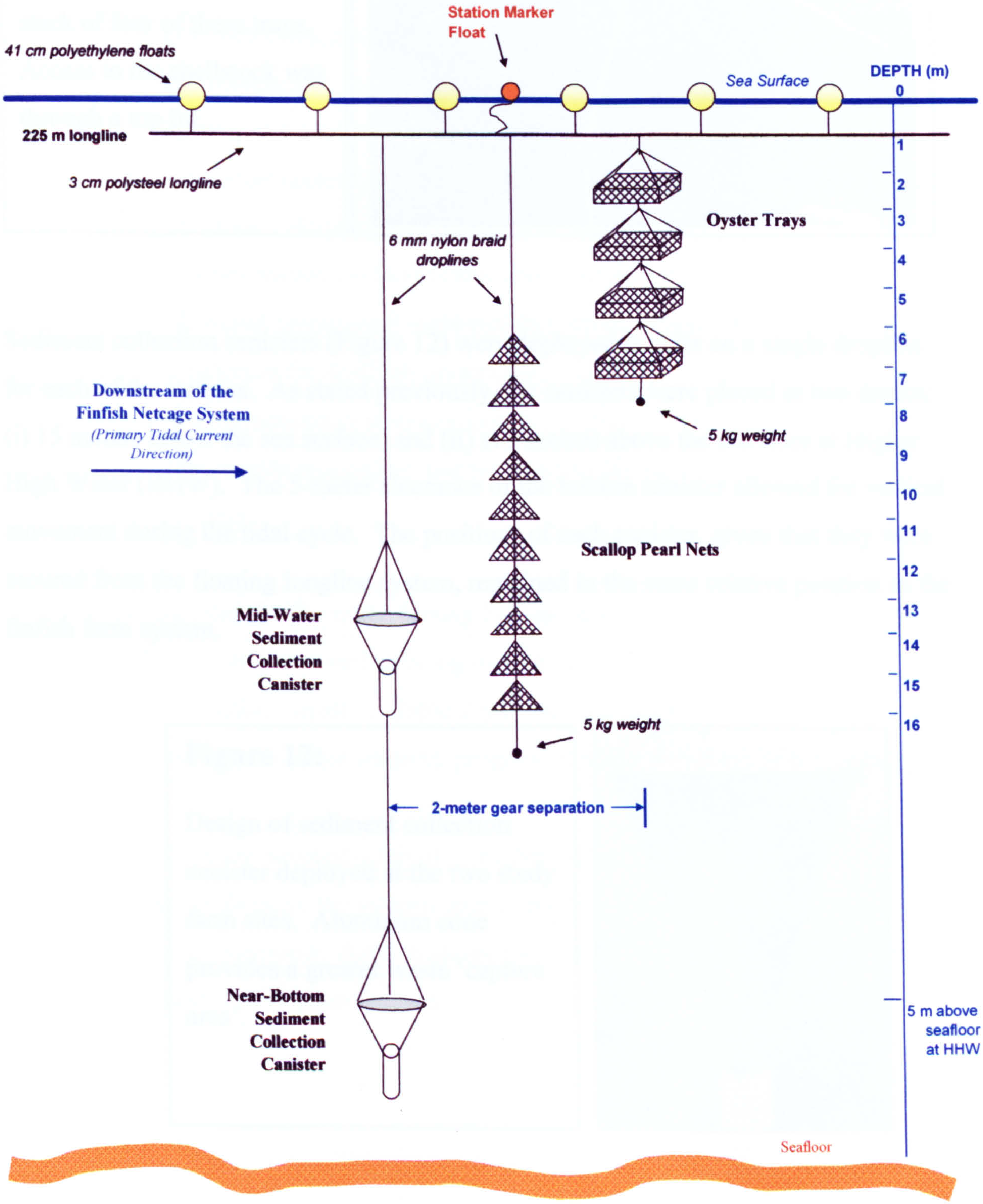
Figure 9: 6 mm scallop Pearl Nets used in study; side and top



The upstream scallop line (closest to the farm system) was initially stocked with larger individuals (5-8 cm). These individuals were routinely sampled (sacrificed) and analysed for contaminant loads during the first half of the study. The second vertical dropline was stocked with scallop seed (approximately 1.0-1.5 cm) and these animals were used for assessing growth performance at the respective study sites. Once the first line was depleted of animals (due to removal for analytical purposes), the scallops on the second line were considered of sufficient size to continue the sacrificial sampling. These scallops were used for both culture performance measures and for analytical sampling until the end of the research program.

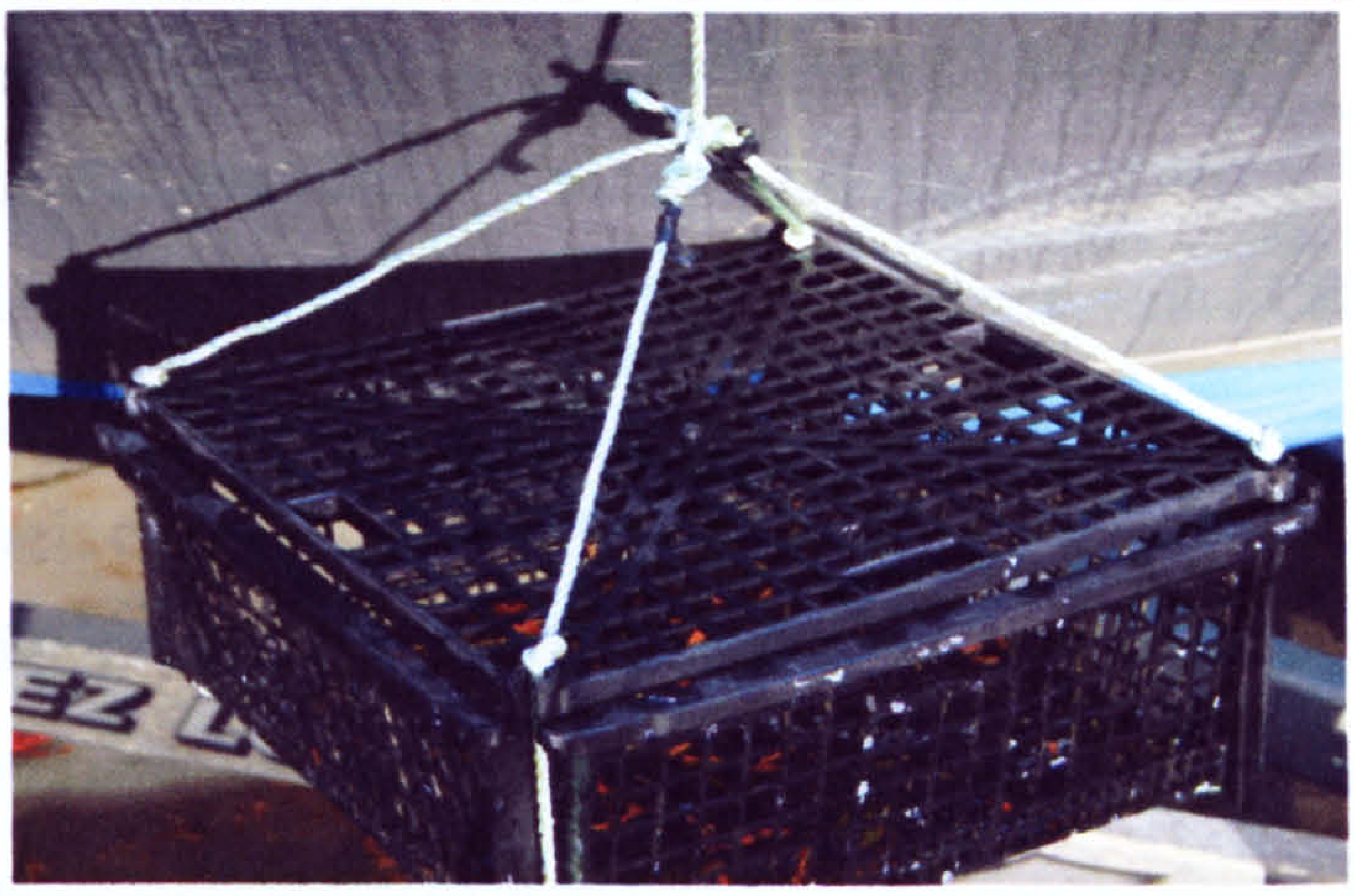
Figure 10

Research infrastructure configuration showing arrangement of shellfish and sediment canister equipment at each sampling station.



A single vertical dropline for oysters was also deployed at each station. A stack of four oyster trays was positioned in the upper layer of the water column, spaced evenly between depths of 1.0 and 6.0 meters. The individual PVC plastic oyster trays (Figure 11) comprise a base of 75 x 75 cm with a vertical dimension of 35 cm (see adjacent photograph).

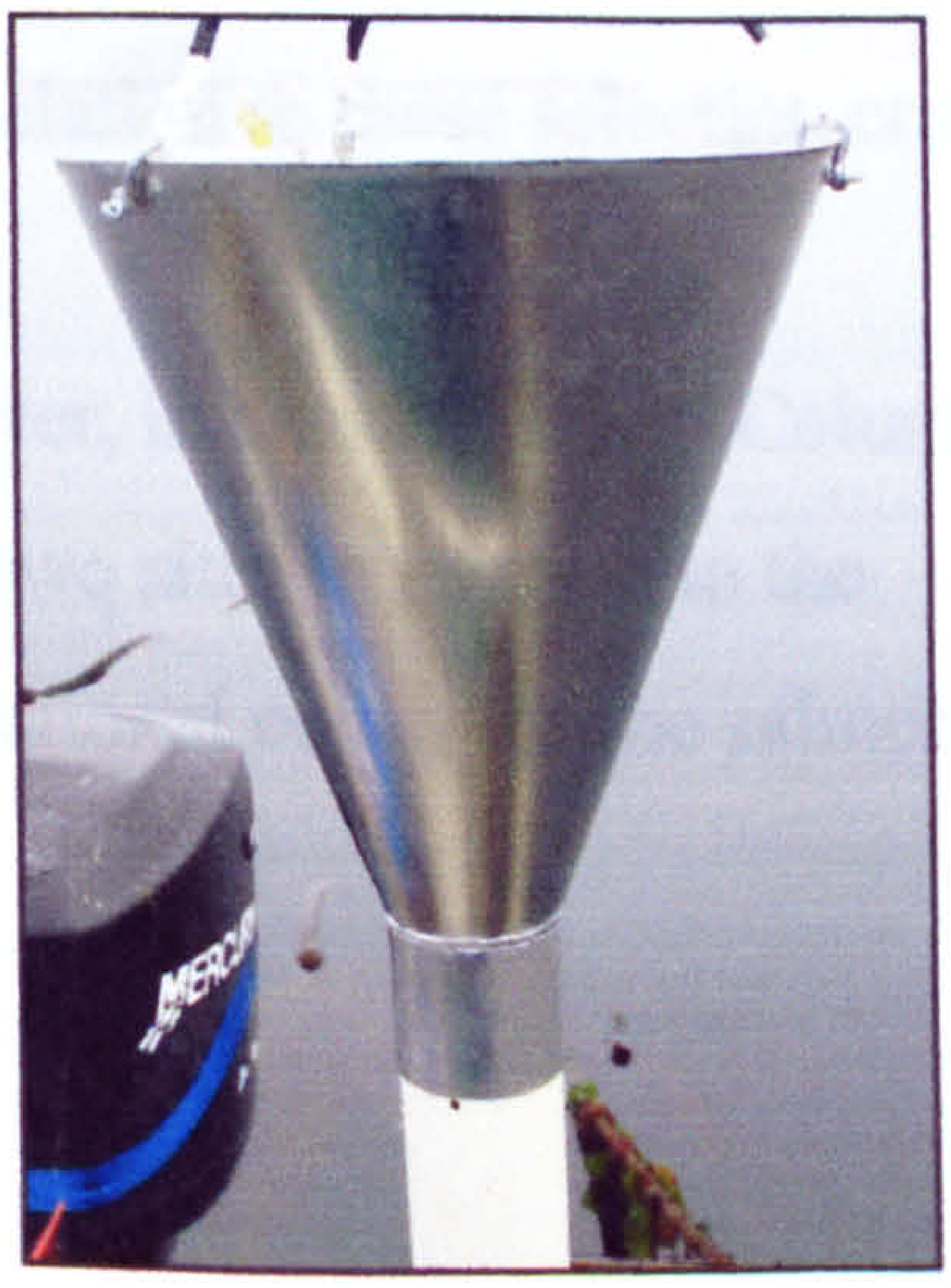
Figure 11: individual oyster tray used in study; each station comprised a stack of four of these trays. Access to the shellstock was through a top lid.



Sediment collection canisters (Figure 12) were deployed in pairs on a single dropline for each of the stations. As stated previously, the canisters were placed at two depths: (i) 15 meters below the sea surface; and (ii) at 5 meters above the sea floor at Higher High Water (HHW). The 5-meter clearance of the bottom canister allowed for vertical movement during the tidal cycle. The positions of each canister, given that they were secured from the floating longline system, remained in the same relative position as the finfish farm system.

Figure 12:

Design of sediment collection canister deployed at the two study farm sites. Aluminum cone provides a greater waste 'capture area'.



2.2 Study Sites

Two marine finfish farms were selected for this study. The criteria used to select these sites included an effort to have sites that:

- would be represented by both an Atlantic salmon (*Salmo salar*) and Pacific salmon (*Oncorhynchus tshawytscha*) production facility, ensuring that unique aspects of the operations that could reflect contaminant loading differences would be assessed (e.g., chemotherepeutant usage, organic waste loading associated with FCR characteristics, etc.);
- had comparable levels of production, ensuring similar waste material loading and potential level of environmental effects;
- were represented by significantly different physical oceanographic properties, allowing an evaluation of waterborne contaminant dispersion and environmental persistence under varying physical conditions;
- displayed biophysical attributes that would suggest the sites were good candidates for shellfish aquaculture; and
- were a reasonable distance from a coastal community center to ensure that sample acquisition and transport to laboratories occurred in a timely manner.

The two sites selected for the study, Young Passage and the Venture Point, satisfied each of the above criteria, as well as being in relatively close proximity to further support survey/sampling logistics. Table 2 provides a summary of the attributes of the farm sites that were used in this research program in relation to these selection criteria.

The two study sites are situated north of Campbell River, in coastal British Columbia, Canada. Figure 13 displays the general area of these two sites in relation to the Vancouver Island community that represents the operational center for the salmon aquaculture industry in western Canada.

Table 2

Comparison of study site attributes in relation to site selection criteria.

Site Selection Criteria	Young Passage	Venture Point
Culture Species	*Oncorhynchus tshawytscha*	*Salmo salar*
Production Level	2,500 MT	2,550 MT
Oceanographic Properties	4.5 meter tidal exchange slow-moderate tidal velocities	4.5 meter tidal exchange moderate-high current velocities
Shellfish Culture Capability	protected waters; very good potential	protected waters; good potential
Distance to Staging Center	24 km	18 km
Proximity of Sites	close; 6.0 km	close; 6.0 km

Approximately 25 salmon farm sites are located in the waters serviced by the center of Campbell River. The other major growing areas are located in the Broughton Archipelago (east of Port Hardy, Figure 12) and the west coast of Vancouver Island, southwest of Courtenay. The industry supports both Atlantic (*Salmo salar*) as well as Pacific (*Oncorhynchus tshawytscha)* salmon aquaculture, with the species generally segregated by latitude. While the Atlantic salmon appears better suited to the colder waters of the north, the Pacific salmon generally out-competes this introduced species where water temperatures are warmer and experience greater annual variation. The sites selected for this study, which examined sites comprised of both species, are located in an area represented by the southern extreme for the Atlantic culture and northern for the Pacific salmon operations.

Figure 14 illustrates the location of the Young Passage and Venture Point study sites in relation to each other. The former is situated off of Nodales Channel, to the north of Sonora Island, while the latter operates along the southern shore of Sonora Island in Okisollo Channel. Both sites are influenced by tidal exchange with waters of Discovery Passage via Johnstone Strait to the west.

Figure 13

General area of finfish aquaculture study sites with respect to coastal population centers of southwestern British Columbia, Canada.

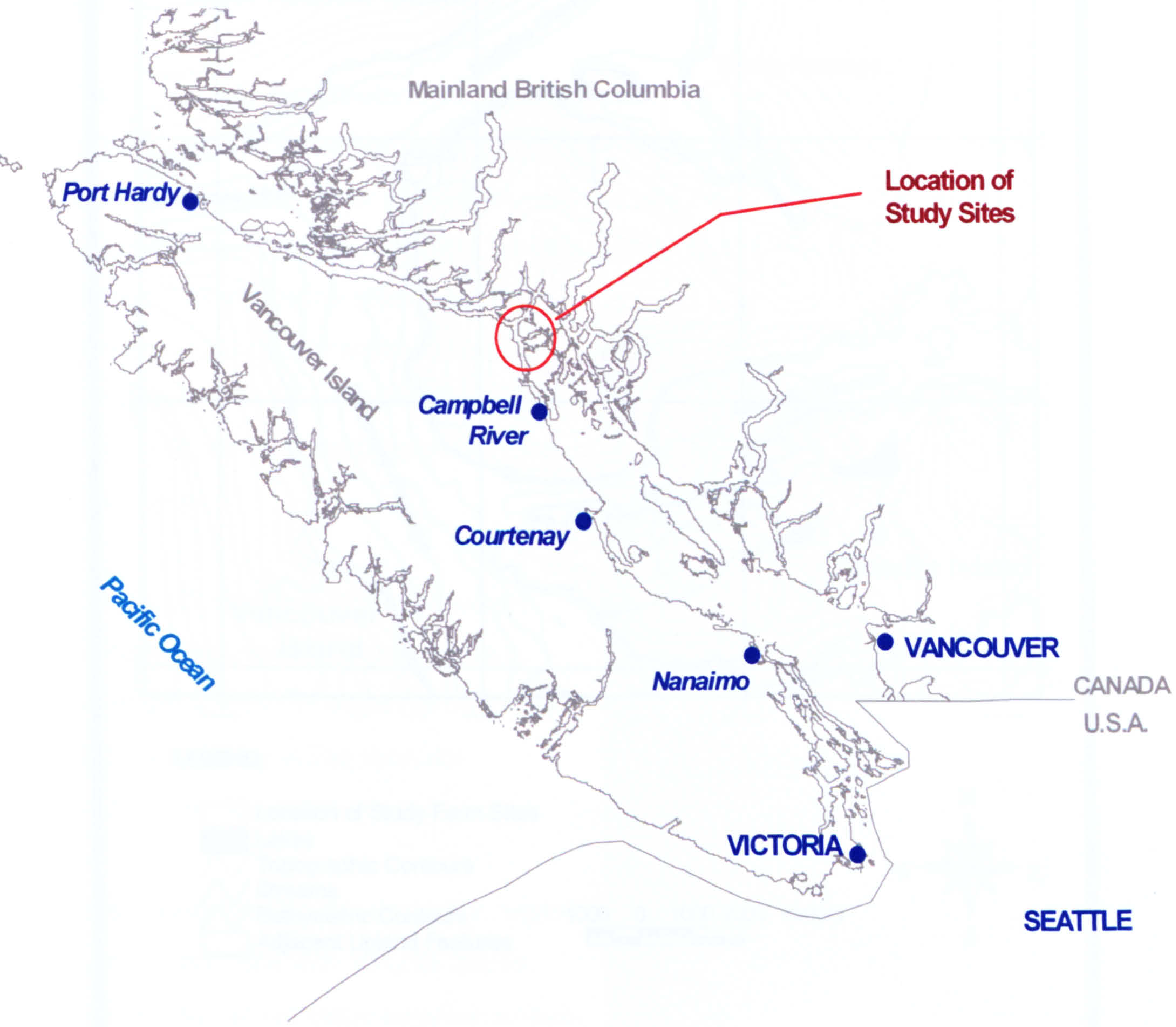
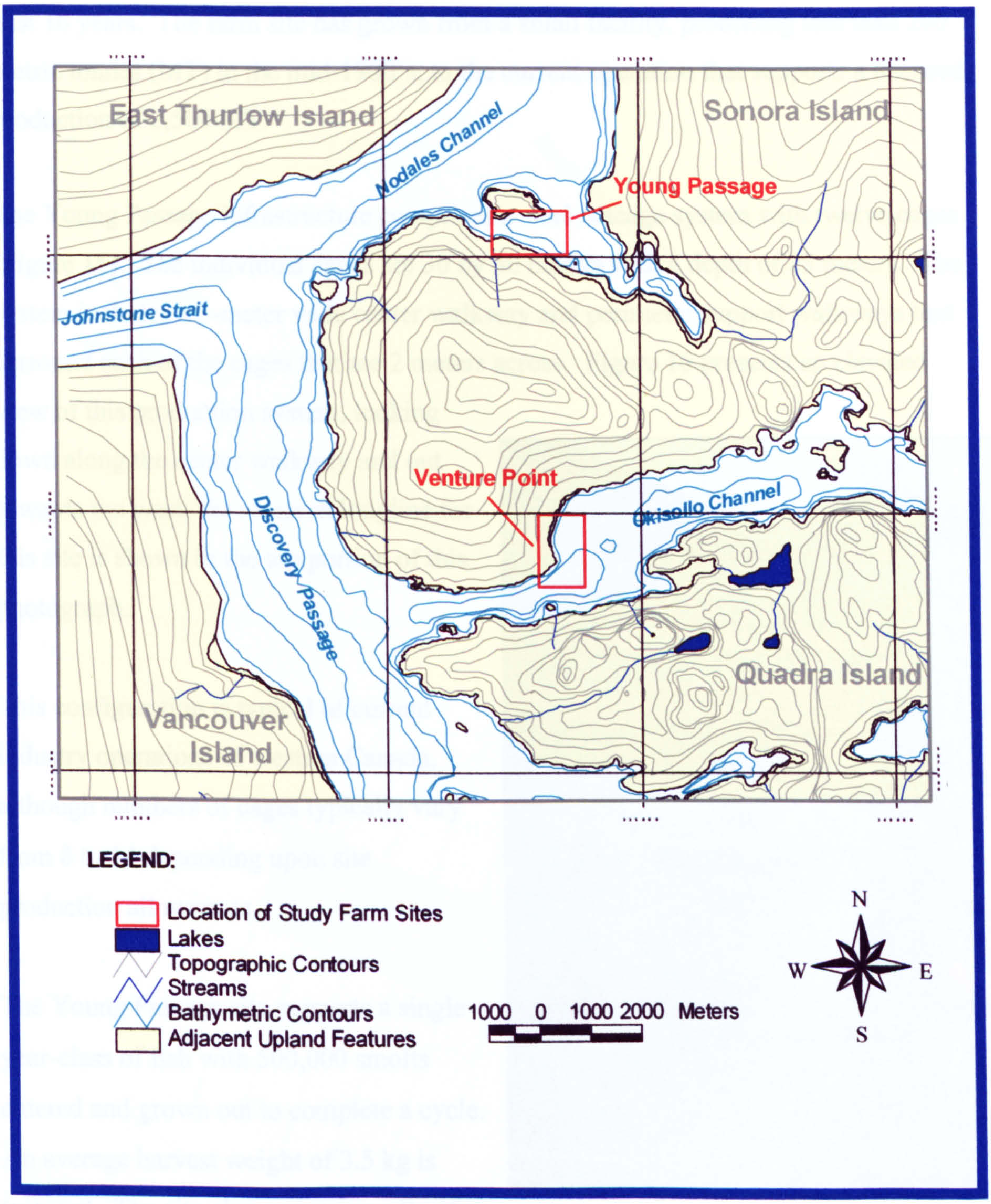


Figure 14

Location of Young Passage and Venture Point study sites



2.2.1 Young Passage Farm Site

The Young Passage farm is a Pacific salmon (*Oncorhynchus tshawytscha)* production site. This site is well established, with a salmon farm operating at this location for the past 16 years. The farm site has grown from a small facility, producing less than 200 metric tonnes (MT) in the mid-1980's, to the current operation that supports a licensed production of 2,500 MT.

The Young Passage infrastructure comprises a steel netcage system with twelve cages (Figure 15). The individual cages are 30 by 30 meters with a depth of 25 meters. The system includes a 3-meter wide center walkway and perimeter support walkways that surround each of the cages that are 2 meters across. Figure 16 presents an elevated view of this production system, looking down along the center walkway and out towards the west; the research longline for this site is shown in the top portion of this photograph.

This configuration is typical of current industry operations of western Canada, although numbers of cages typically vary from 8 to 14 depending upon site production allowances.

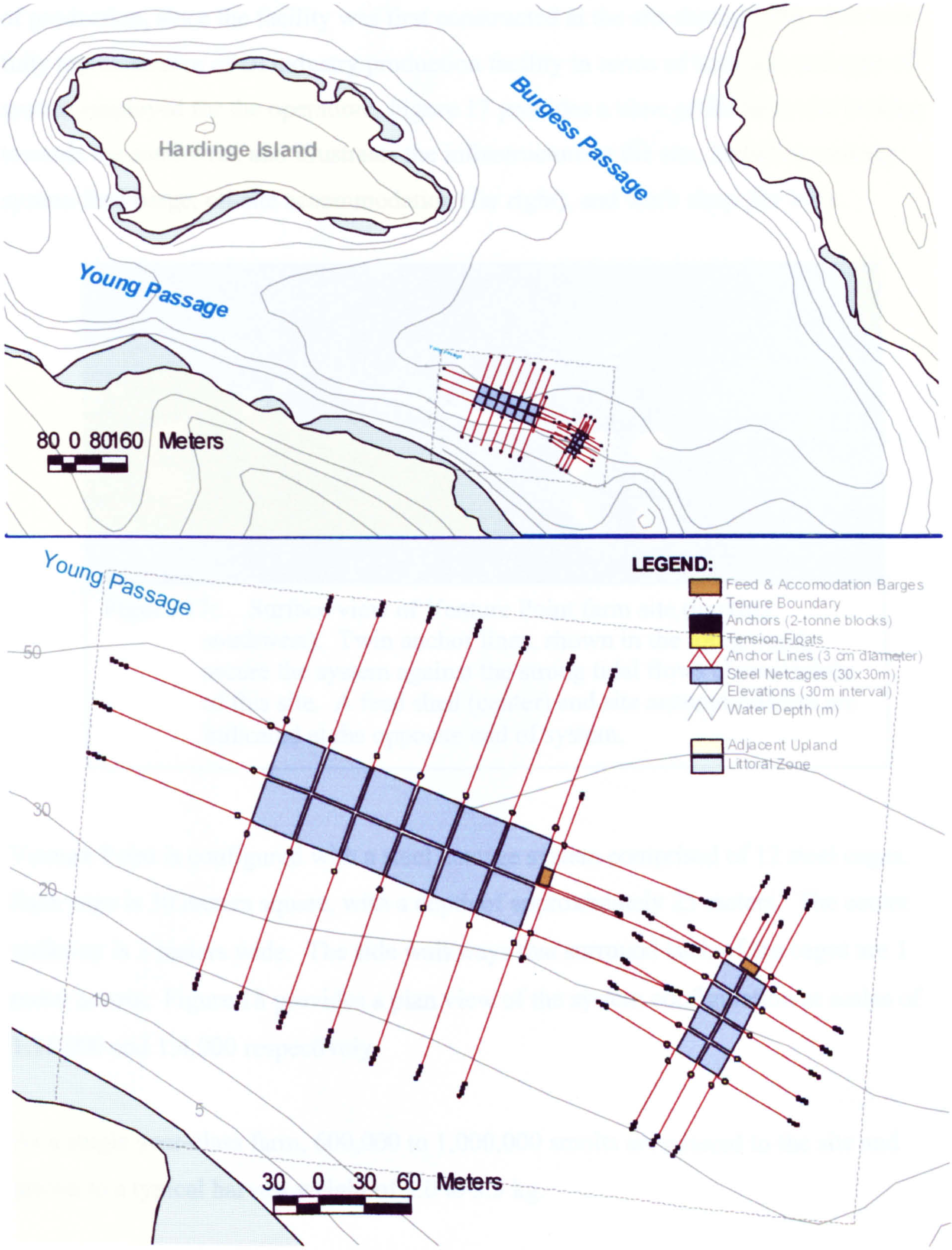
The Young Passage site supports a single year-class of fish with 500,000 smolts entered and grown out to complete a cycle. An average harvest weight of 3.5 kg is targeted. The site also maintains a Pacific salmon broodstock system to the southeast of the production cages (Figure 15).

Figure 16 Elevated view of Young Passage farm site.

Figure 15

Farm operational configuration at Young Passage study site.

Top Map 1:15,000 Lower Map 1:4,000



2.2.2 Venture Point Farm Site

The Venture Point site is an Atlantic salmon (*Salmo salar*) farm site. The farm has been operating for 15 years and is currently licensed to produce 3,000 MT of fish. As with the Young Passage farm site, this farm operation has continued to grow, in terms of production, since the facility was first constructed at the site during 1988. It is now fully representative of an industry production facility in terms of both size and type of system employed for the operation. Figure 17 provides a view of the farm site looking towards the southwest, and illustrates the infrastructure at the site, including netcage system feed barge, on-site accommodation (far right), and work shop (far left)..

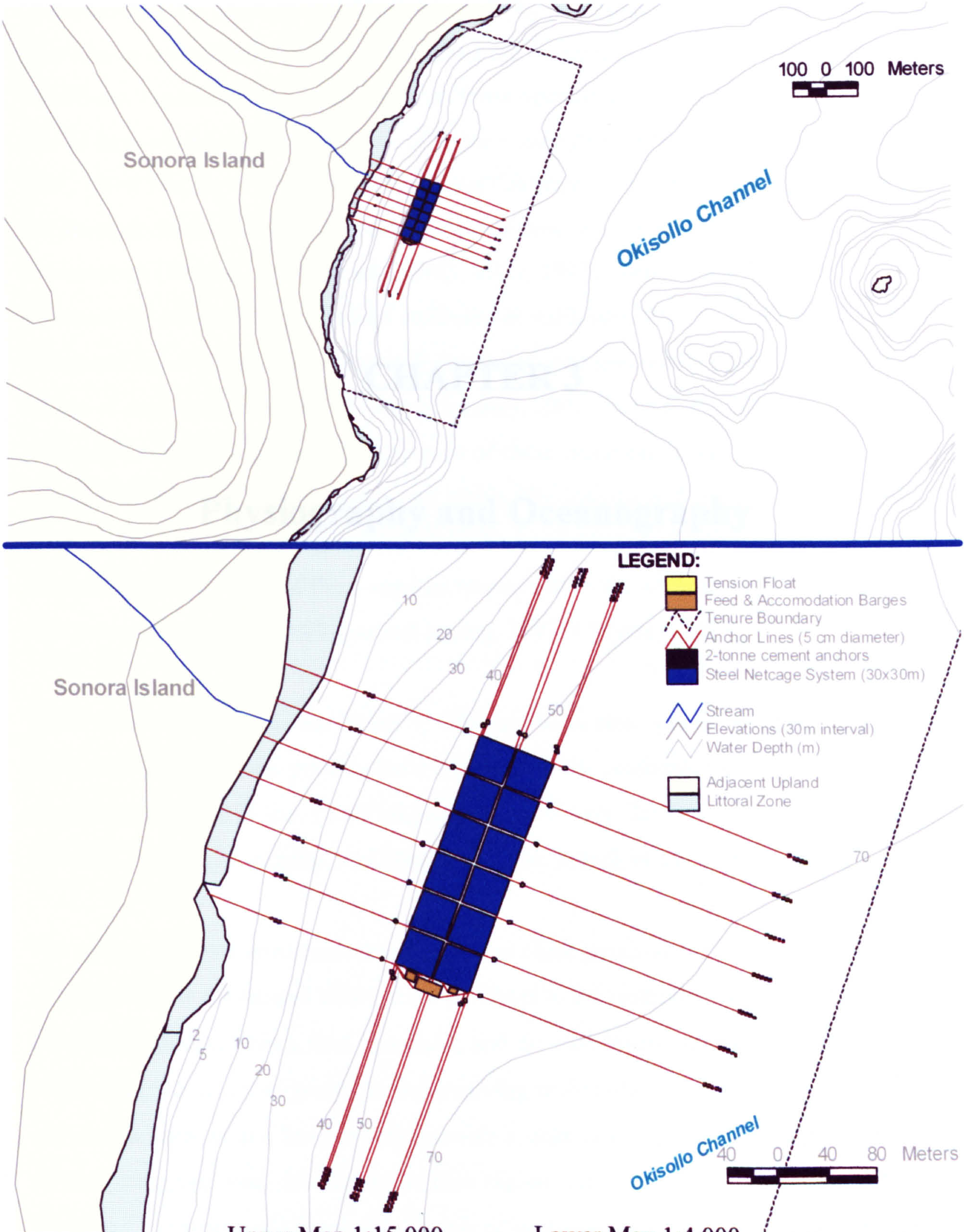
Figure 17: Surface view of Venture Point farm site (looking southwest). Twin anchor lines, shown in the foreground, secure the system against the strong tidal flows characteristic of this site. A feed shed (center) and site accommodation are indicated at the opposite end of system.

Venture Point is configured with a steel netcage system comprised of 12 steel cages. Each cage is 30 meters square, with a depth of approximately 25 meters. The center walkway is 3 meters wide. The side walkways that surround each of the cages are 1 meter across. Figure 18 provides a plan view of the system configuration at scales of 1:15,000 and 1:4,000 respectively.

As a single year-class farm, 600,000 to 1,000,000 smolts are entered to the site and grown to a typical harvest weight of 3.0 to 3.5 kg.

Figure 18

Farm operational configuration at Venture Point study site



Upper Map 1:15,000 Lower Map 1:4,000

CHAPTER 3

Physiography and Oceanography

3.1 Introduction

Understanding the physical oceanographic and physiographic characteristics of finfish farm sites is considered essential to assessing the dispersion processes of the solid and waterborne contaminants originating from these operations. Although numerous studies have examined the dispersion of waste materials from farm operations (Gowen et al., 1994; Findlay et al., 1994; Wildish and Cranston, 1997; Brooks, 2002), as well as the physical-chemical and biological impacts of these wastes to the benthic environment (Weston, 1990; Brooks, 2002; EAO, 1997), surprisingly few have concurrently assessed these physical attributes in sufficient detail as to allow a thorough understanding of the processes that will determine, and potentially predict, the fate and effects of such waste material loading (Chromey, 2001; Perez et al., 2002). Researchers fully recognize the importance of these processes, suggesting that the spatial distribution of organic wastes are highly site-specific and rely not only on tidal currents, but on bottom topography, flocculation and erosion, all of which will influence residency time of these organic wastes within the water column (Sutherland, 2001) or on the seafloor (Milligan and Loring, 1997; Chromey, 2001).

Water circulation patterns are driven by tidal and local atmospheric processes, although the former represents the primary force that defines the predictable periodicity in flow conditions at any farm site. Local tidal energies will vary, depending upon position on the earth, local physiography, and time of the year, month or day (Gross, 1977).

The pattern of water movement through, and in close proximity of a marine net cage system provides substantial information pertinent to the understanding of waste material dispersion from a farm operation, and the subsequent ecological consequences of this material as it is dispersed in the receiving environment. Measurement of water *circulation* patterns at a farm site can provide a large-scale picture of how the water moves through an area, describing the flow regime that is produced by the physical, disruptive influences of topography (points of lands, embayments), bathymetry (bottom slope, complexity), and interference effects of the aquaculture infrastructure (cages, feed sheds, anchors and anchor lines).

These physiographic characteristics play an important role in defining the hydrodynamic influences at an aquaculture facility (Cross, 1993; Cross et al., 2001), the results of which are very site-specific. For example, water flow around a point of land may create back-eddy conditions that could entrain particulate wastes and result in localized accumulation areas for such solids. In contrast, a prominent bathymetric feature (such as a reef) may disrupt flow in such a way as to cause localized upwelling and thus re-suspension of organic waste particulates, resulting in significantly greater dispersion of this material.

The environmental fate and effects of organic wastes from a finfish net cage would also be anticipated to vary as the intensity, or magnitude of these physical processes varied. The velocity and duration of water flow through a cage system will determine potential dispersion, dilution, and eventual concentration of these wastes once they are released into the water column. Higher flows will typically result in a greater dispersion, while slower (or minimal) flow conditions will result in localized (and highly concentrated) waste accumulations.

Water flow is also an important factor in the process of release (dissolving) of nutrients from suspended particulates during the waste dispersion process (Sutherland, 2001). Physical agitation, abrasion, and continual exposure of the surface area of these particulates to oxidation within a turbulent water column will increase the rate of this assimilative process, resulting in a reduced depositional component over that of a dissolved fraction.

The implications of maintaining adequate flows through an aquaculture facility results in a number of environmental benefits, as well as numerous economic ones in view of the potential offered through consideration of an integrated, or Multi-Trophic Aquaculture (MTA) approach to aquatic farming. Sustained water flows will ensure adequate exchange and maintenance of water quality conditions (nutrient flux, oxygen concentrations, clarity) required for the health of the culture species, whether finfish, shellfish, or macrophytes. Adequate flow will also affect husbandry practices, allowing (to some degree, and dependent on species) an increase in stocking density, an important consideration in the design of a commercially viable MTA system.

As discussed above, the release of organic wastes will be processed and environmentally partitioned (benthic versus water column), as a result of a combination of physical and chemical assimilative processes (Brooks, 2001). The water circulation characteristics of a site, including tidal velocities, quiescent periods, stratification, and turbulence, will determine the rate of these assimilative processes and as such define how these wastes are partitioned, and hence their respective bioavailability to the proposed culture components (species) of an MTA system.

The importance of oceanographic and physiographic characteristics to site selection in terms of culture performance, whether monoculture or MTA, is clear. However, the processes by which organic wastes are degraded and partitioned, environmentally, will also affect the partitioning and potential bioavailability of any trace-level contaminants associated with these wastes. The chemical constituents of feed that are presumably released in fish faeces comprise a variety of compounds, including those associated with antibiotic treatments, trace metal micronutrients (e.g., zinc), and elemental sulpher. These constituents have been measured in sediments (Johnsen et al., 1993; McGhie et al., 2000; Sutherland et al., 2002; Yeats, 2002) and have been suggested as possible tracers of farm wastes for assessing far-field dispersion and the potential ecological effects of this material.

Documentation of the oceanographic and physiographic characteristics of the study sites selected for this research program represents a key component in the assessment of waterborne, bioavailable contaminants, and their possible risks to seafood safety in the context of developing Multi-Trophic Aquaculture (MTA) systems. The evaluation of waste dispersion pathways and processes that specifically relate to the integration of shellfish (bioaccumulators) with finfish provides a probable 'worse-case' scenario for such a system development.

3.2 Objectives

A suite of physical oceanographic assessments were used to determine the primary dispersion pathway, or downstream vector, extending from each of the study farms, as well as to document the spatial and temporal patterns of water flow that affect waste dispersion and environmental persistence of waterborne contaminants. The specific objectives of this research component included completion of:

- an initial hydrographic evaluation of each study farm site to document the primary contaminant dispersion pathway, thereby identifying the appropriate corridor for the experimental infrastructure deployments;
- a tidal circulation survey to document the spatial pattern of water movement through the study sites (both horizontal and vertical components);
- a detailed evaluation of lunar-cycle tidal flow dynamics at a fixed point along the primary contaminant dispersion pathway, including the vertical component (upwelling/down-welling);
- a summary of water column properties over the duration of the study (temperature-salinity profiles); and a
- bathymetric and substrate characterization survey to document the physiographic aspects (marine) of the study sites in relation to the local topographic features.

The results of these surveys are subsequently discussed in terms of the site characteristics that will influence contaminant dilution, dispersion, persistence as well as those attributes that will affect shellfish growth and survival in consideration of an integrated finfish-shellfish aquaculture system, and the development of Multi-Trophic Aquaculture (MTA) systems in general.

3.3 Materials and Methods

The oceanographic and physiographic characteristics of the two study sites were assessed using a combination of remote data acquisition systems, providing information on hydrodynamics, water column density (temperature-salinity profiling), bathymetry, and substrate classification. The field survey, data processing, and results presentation methodology for these study components are provided in the following sections.

3.3.1 Tidal Influences

Two types of tidal current measurements were completed during this study, including: (i) an evaluation of tidal exchange dynamics over a complete lunar cycle; and (ii) a tidal circulation survey of the farm site area. Jointly these physical oceanographic measurements provide a comprehensive assessment of water column changes that will have a direct influence on potential waterborne contaminant dilution, dispersion and persistence in the vicinity of the farm sites.

The first data set provides a detailed evaluation of the temporal change in tidal activity at the site. These data illustrate the tidal dynamics as they relate to the lunar cycle (M2 tidal constituent), and allow an evaluation of the magnitude and direction of flow over time. In this assessment, these data can help define residual flow, maximum and average velocities over the cycle, and duration of periods when flows are minimal (quiescent time) and when they are of a magnitude that re-suspension processes may be in effect at a farm site.

The second data set provides a snapshot of the flood and ebb currents, over the entire area of the site, and over most of the water column (vertical and horizontal flow patterns). These spatial data best illustrate the presence, size, and intensity of eddies, as well as documenting other flow structure (e.g., upwelling or down-welling) associated with complicated bathymetry and topography typical of coastal farm sites.

Tidal Cycle Dynamics: ***Instrument Deployment***

These data provide tidal current data at one specific (presumably representative) position only, but over an entire lunar cycle. These time-series data show the variations in tidal currents over the spring/neap cycle, as well as any effects due to wind, estuarine circulation, etc.

As an initial site assessment, during September 2001, a pair of SD-6000 tidal current meters was deployed at each study site to roughly establish the residual flows for the sites and thus to provide the information upon which the position of the experimental shellfish longline could be determined. These two meters, placed at 15 meters below the surface (MLLW) and 5 meters above the seafloor (30-minute sampling interval) provided preliminary site data only, and a more detailed assessment of tidal cycle dynamics was provided through a subsequent instrument deployment as described below.

At each study farm the RDI Broadband (600 kHz) Acoustic Doppler Current Profiler (ADCP) was moored on bottom (directed, or "looking" upwards), with current speed-direction data averaged within 2-meter "bins" from near-bottom to near-surface. Mounted near the sea bottom, the 300 khz ADCP unit provides precise measurements of ocean currents (both the horizontal and vertical components) at many levels within the water column in water depths up to approximately 75 meters.

These acoustic Doppler instruments measure particle velocity by detecting the Doppler shift in acoustic frequency, arising from water current movements, of the backscattered returns of upward (20° from vertical) transmitted acoustic pulses. The Doppler shift of the 600 kHz acoustic signal is used to determine water velocities at a vertical spacing of 2 meters. The sampling scheme for this study involved 10-minute ensembles, composed of 100 water-column pings spaced evenly over the sampling interval (ensemble), or every 6 seconds. The estimated accuracy of each velocity measurement is ±1.0 cm/s (RD Instruments).

Tidal Cycle Dynamics: *Data Processing*

The acquired ADCP data contained velocity data every 2 meters in the vertical (entire water column). Five depth plane bins from each site record were selected for analysis, representative of near surface (5 metres), mid-cage system and lower region of the shellfish growing area (15 metres), bottom of the finfish netcage system (25 metres), and approximately 10 metres above the seafloor (~35 metres from the surface). These data were corrected for magnetic deviation, times converted from local to UTC, and then converted to an ASCII format for processing. Post-processing steps included:

The initial data processing step comprised a removal of "spikes", resulting in a data file that excluded points that were considered outliers in terms of magnitude of the reported velocity. The resulting data were analysed in a number of ways, and results displayed in a series of summary graphs and tables for each study site. Current speed and direction tables/plots, tidal current ellipse plots, and progressive vector diagrams were used to examine and discuss apparent trends within these data.

Tidal Circulation Survey: ***Field Approach***

In addition to acquiring tidal current data from the fixed mooring, an ADCP circulation study was completed for each of the study sites. The RDI Sentinel Workhorse (300 khz) ADCP was mounted alongside a survey vessel that also supported the associated computing and electronic positioning equipment. The ADCP instrument extended into the water column approximately 0.5 meters, allowing data to be acquired from a depth of approximately 3.0 meters to that of just above the seafloor. The ADCP has a blanking distance of approximately 2.5 meters in front of the transducer assembly and thus the near-surface flows can not be assessed using this particular instrument (a higher frequency Doppler can assess flows in smaller depth bins, with a reduced blanking distance, but these instruments have an associated depth limitation).

Just prior to the maximum flows of selected spring ebb and flood tides of the survey day, the vessel was navigated along a pre-determined path (vessel speed 1.5-2.5 kts), providing the opportunity for the instrument to acquire current velocity and direction information from the entire water column (accounting for measured vessel speed over the sea floor).

Tidal Circulation Survey: ***Data Processing/Analysis***

Data acquired from this circulation survey were compiled in formats specific to the instrument application, i.e., raw/processed data files of the acoustic/spatial information derived from the ADCP over the survey period. The raw data acquired from the instrument were subjected to a de-spiking procedure as an initial step in the data processing. Secondly, the de-spiked current speed-direction data were bin-averaged within the horizontal and the vertical axes. In the horizontal axis *average* current speed and direction data were produced using the raw data collected over each 20-meter interval traveled along the vessel track. Vertical data were similarly summarized (within 2-metre depth bins) and both were compiled into an ASCII-formatted processed data file that also contained bathymetric data (for each transducer), as well as vessel tracking information (distance east and north from the starting position). The geographical starting position for each survey was acquired using a differential Geographical Positioning System (12-channel ML-250 GPS) providing a survey positional accuracy of ± 3 meters.

For each farm site, an appropriately scaled map base was developed using ARC/VIEW 3.1 Geographical Information System (GIS) software. The tidal circulation data, for representative depth planes (flood and ebb tide scenarios), were printed directly onto the GIS base maps. The location of farm structures and the shellfish monitoring infrastructure was also depicted on the site base map, thereby allowing a visual assessment of the tidal circulation pattern, depth contours and position/alignment of farm structures, concurrently.

3.3.2 Bathymetry and Substrate Composition

A bathymetric and substrate classification survey was conducted for each of the study sites using a Questar-Tangent QTC-Profiler. This system post-processes the back-scatter acoustic signal generated from a high resolution depth sounder (50 khz), and correlates the signal pattern with seafloor penetration depth, thereby allowing a remote substrate classification of the seafloor.

The site surveys were completed from a 7-meter surface vessel equipped with Suzuki ES-2025 color video sounder (frequency: 50 khz), a ML-350 differential-GPS, and the QTC signal processing unit. Data from each instrument were concurrently logged onto an onboard computer, providing a continuous stream of bathymetric (depth soundings), raw QTC acoustic back-scatter data, and the associated geographic position of the survey vessel determined by the GPS.

QTC-Impact software was used to define the substrate classification categories (types) of the raw acoustic back-scatter data, employing a combination of Principal Components Analysis (PCA) and hierarchical cluster analysis to statistically differentiate groups of bottom types based on acoustic echo. The geo-referenced classification assignments and depth data were used to create colour contour plots (plan view; Surfer 7.0 software) for each study site. The six back-scatter (intensity) categories, derived from the soundings, resulted in a substrate gradient comprised of six categories ranging from hard (rock) through soft (silt-clay) fractions. The gradients were specific to each study site and were not calibrated to provide a direct comparison between these survey results.

A series of eight 0.1 m^2 Van Veen grab samples were acquired systematically from each of the study areas, spanning each of the estimated substrate types identified by the QTC profiler, to validate substrate classification results. In areas where grab samples proved difficult, a Seamor Remote Operated Vehicle (ROV) was deployed to examine these substrates.

3.3.3 Temperature and Salinity Profiles

Temperature and salinity (T-S) profiles were completed at each of the study sites by farm site staff. These data were acquired weekly using electronic meters with direct surface read-out. Data were recorded in the Daily Site Logs of the farm, and transferred to a corporate database that compiled operational information for all of the company's farm sites.

Temperature and salinity data specific to the period of study were extracted from the corporate database and compiled into a working file for each farm site. These data were initially examined for potential outliers, and suspect data removed from the working files. Simple scatter plots of temperature over time were presented for each site, while salinity data were summarized as mean, maximum and minimum values over this survey period.

3.4 Results

The oceanographic and physiographic characteristics of each study site are presented in a series of summary figures that outline important trends and inter-site differences important for the assessment of the finfish-shellfish integrated aquaculture approach and the potential for waterborne contaminant influences on the shellfish component of such a system.

3.4.1 Tidal Cycle Dynamics

Tidal current results for the fixed mooring deployment are summarized and presented for each of the farm sites, independently. A comparative evaluation of these results is presented in the Discussion, with a dialogue of the implications of these attributes on the other aspects of this research program (Chapters 4,5,6, and 7).

Venture Point Farm Site:

The tidal current speed and direction data were acquired from a 600 khz ADCP moored at the Venture Point farm site from 25 September 2003 through 27 October 2003, providing a total of 4,599 records (10-minutes sampling interval). Table 3 presents the summary statistics for each depth plane assessed from the vertical profile acquired through the deployment.

Average velocities range from 12.32 cm/s near the bottom to 14.18 cm/s near the sea surface. Maximum velocity achieved over the sampling period typically exceeded a knot, with flows of between 49 and 54 cm/s. The vector-averaged current speeds at this site show an almost uniform flow through the water column, with a net (residual) flow to the southwest. The experimental shellfish longline was place along this dispersion vector.

Table 3: Summary statistics for tidal current meter (ADCP 600 khz RDI Broadband) deployed at the Venture Point farm site from 25 September through 27 October 2003. Sampling interval of 10 minutes; n= 4,599. Four depth planes selected for analysis: 5, 15, 25, and 35 meters below sea level (MLLW).

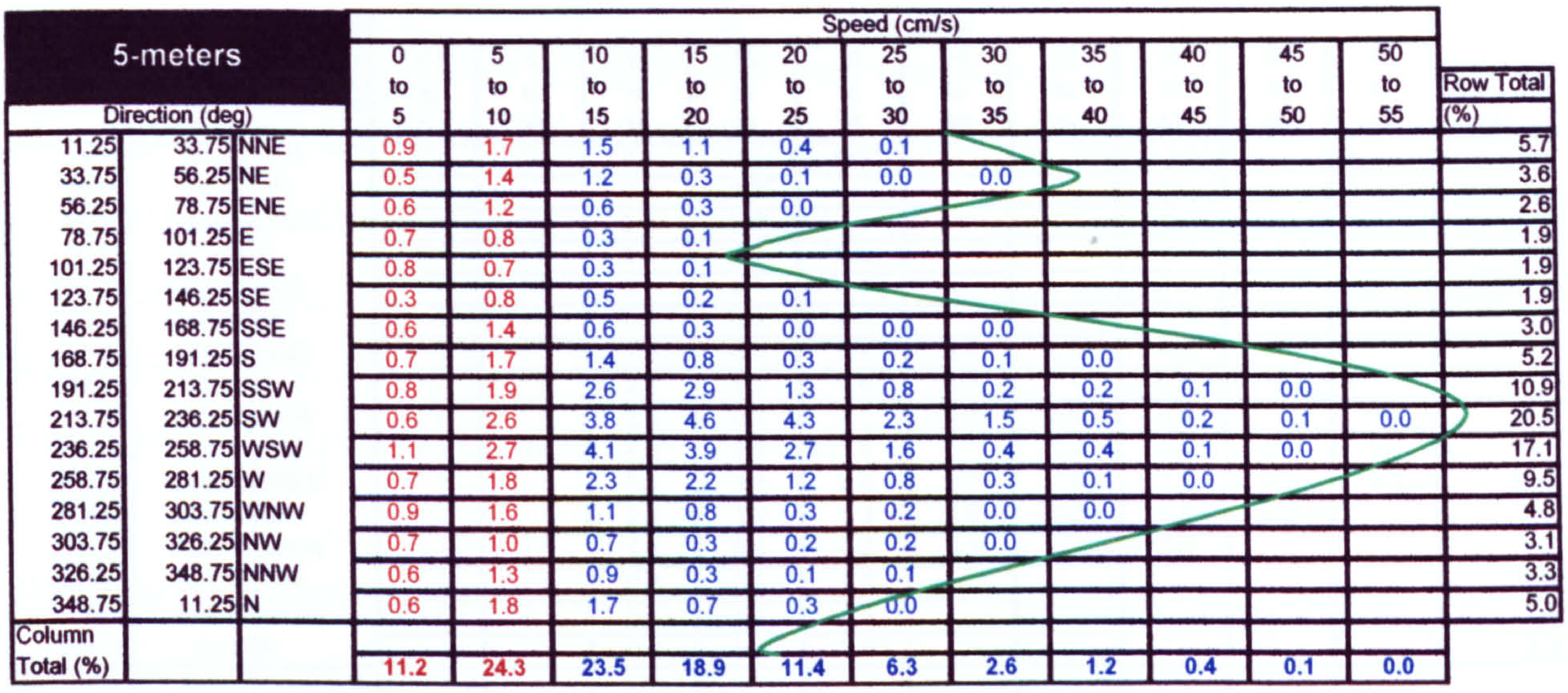
Depth Plane	**Mean Velocity** *cm/s*	**Maximum Velocity** *cm/s*	**Vector Averaged Speed** *cm/s @ deg. T*	
5 meters	14.18	52.11	8.28	239.30
15 meters	13.38	49.74	8.64	237.00
25 meters	12.51	49.16	8.37	232.60
35 meters	12.32	53.40	7.70	233.10

Tidal current data for the Venture Point farm site are further analysed through a series of summary tables (Tables 4, 5) and plots (Figures 19, 20 and 21).

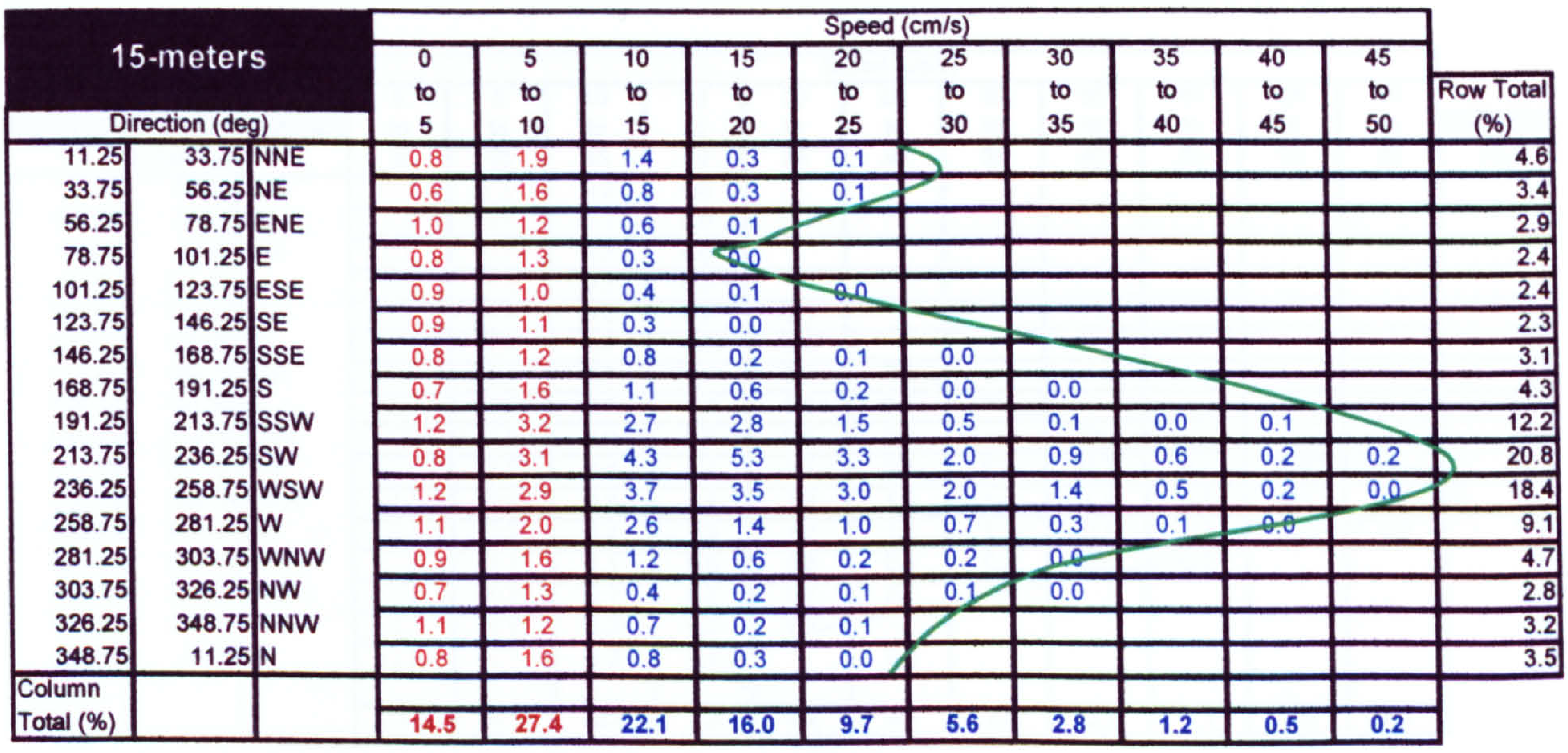
Tables 4 and 5 present Speed versus Direction frequency tables for the 5- and 15-meter depth planes, and the 25- and 35-meter depth planes, respectively. These tables provide a breakdown of the tidal currents into speed and direction bins; a bin size is 5 cm/s by 22.5° (True) was used in this data summary. The green lines illustrate the change in direction over the tidal cycle.

An estimate of the frequency of 'slow' tidal flows are provided by indicating those periods where currents are <10 cm/s (red values), which presumably represent conditions under which minimal dispersion or re-suspension of organic wastes might occur. In contrast, the periods of the tidal cycle where flows are >10 cm/s (blue values) have, generically, been suggested as a velocity threshold beyond which particulate suspension and/or re-suspension processes could influence the dispersion patterns of organic wastes from a farm site. This threshold has been employed in mathematical models of waste dispersion to trigger algorithms that describe the re-suspension process (Chromey, 2001).

Table 4: Venture Point 600-khz ADCP tidal data acquired 25 September 2003 through 27 October 2003. Speed-direction frequency table for 5 meter and 15 meter depth bins. Directional trend over deployment cycle indicated with green line. Blue values suggest speeds at which potential suspension and re-suspension processes may occur. The red values area considered to represent periods within which depositional processes may occur (*quiet time*).



5-meters			Speed (cm/s)											
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	50 to 55	Row Total (%)
11.25	33.75	NNE	0.9	1.7	1.5	1.1	0.4	0.1						5.7
33.75	56.25	NE	0.5	1.4	1.2	0.3	0.1	0.0	0.0					3.6
56.25	78.75	ENE	0.6	1.2	0.6	0.3	0.0							2.6
78.75	101.25	E	0.7	0.8	0.3	0.1								1.9
101.25	123.75	ESE	0.8	0.7	0.3	0.1								1.9
123.75	146.25	SE	0.3	0.8	0.5	0.2	0.1							1.9
146.25	168.75	SSE	0.6	1.4	0.6	0.3	0.0	0.0	0.0					3.0
168.75	191.25	S	0.7	1.7	1.4	0.8	0.3	0.2	0.1	0.0				5.2
191.25	213.75	SSW	0.8	1.9	2.6	2.9	1.3	0.8	0.2	0.2	0.1	0.0		10.9
213.75	236.25	SW	0.6	2.6	3.8	4.6	4.3	2.3	1.5	0.5	0.2	0.1	0.0	20.5
236.25	258.75	WSW	1.1	2.7	4.1	3.9	2.7	1.6	0.4	0.4	0.1	0.0		17.1
258.75	281.25	W	0.7	1.8	2.3	2.2	1.2	0.8	0.3	0.1	0.0			9.5
281.25	303.75	WNW	0.9	1.6	1.1	0.8	0.3	0.2	0.0	0.0				4.8
303.75	326.25	NW	0.7	1.0	0.7	0.3	0.2	0.2	0.0					3.1
326.25	348.75	NNW	0.6	1.3	0.9	0.3	0.1	0.1						3.3
348.75	11.25	N	0.6	1.8	1.7	0.7	0.3	0.0						5.0
Column Total (%)			**11.2**	**24.3**	**23.5**	**18.9**	**11.4**	**6.3**	**2.6**	**1.2**	**0.4**	**0.1**	**0.0**	



15-meters			Speed (cm/s)										
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	Row Total (%)
11.25	33.75	NNE	0.8	1.9	1.4	0.3	0.1						4.6
33.75	56.25	NE	0.6	1.6	0.8	0.3	0.1						3.4
56.25	78.75	ENE	1.0	1.2	0.6	0.1							2.9
78.75	101.25	E	0.8	1.3	0.3	0.0							2.4
101.25	123.75	ESE	0.9	1.0	0.4	0.1	0.0						2.4
123.75	146.25	SE	0.9	1.1	0.3	0.0							2.3
146.25	168.75	SSE	0.8	1.2	0.8	0.2	0.1	0.0					3.1
168.75	191.25	S	0.7	1.6	1.1	0.6	0.2	0.0	0.0				4.3
191.25	213.75	SSW	1.2	3.2	2.7	2.8	1.5	0.5	0.1	0.0	0.1		12.2
213.75	236.25	SW	0.8	3.1	4.3	5.3	3.3	2.0	0.9	0.6	0.2	0.2	20.8
236.25	258.75	WSW	1.2	2.9	3.7	3.5	3.0	2.0	1.4	0.5	0.2	0.0	18.4
258.75	281.25	W	1.1	2.0	2.6	1.4	1.0	0.7	0.3	0.1	0.0		9.1
281.25	303.75	WNW	0.9	1.6	1.2	0.6	0.2	0.2	0.0				4.7
303.75	326.25	NW	0.7	1.3	0.4	0.2	0.1	0.1	0.0				2.8
326.25	348.75	NNW	1.1	1.2	0.7	0.2	0.1						3.2
348.75	11.25	N	0.8	1.6	0.8	0.3	0.0						3.5
Column Total (%)			**14.5**	**27.4**	**22.1**	**16.0**	**9.7**	**5.6**	**2.8**	**1.2**	**0.5**	**0.2**	

Table 5: Venture Point 600-khz ADCP tidal data acquired 25 September 2003 through 27 October 2003. Speed-direction frequency table for 25 meter and 35 meter depth bins. Directional trend over deployment cycle indicated with green line. Blue values suggest speeds at which potential suspension and re-suspension processes may occur. The red values area considered to represent periods within which depositional processes may occur (*quiet time*).

25-meters			Speed (cm/s)										
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	Row Total (%)
11.25	33.75	NNE	1.2	1.4	0.4	0.1	0.0						3.1
33.75	56.25	NE	0.9	1.3	0.4	0.0							2.7
56.25	78.75	ENE	0.8	1.1	0.3	0.0							2.4
78.75	101.25	E	0.7	1.0	0.4	0.0	0.0						2.2
101.25	123.75	ESE	1.1	1.5	0.5	0.1	0.0						3.2
123.75	146.25	SE	1.4	1.2	0.6	0.2	0.1						3.5
146.25	168.75	SSE	1.1	2.1	1.2	0.5	0.2	0.0	0.0				5.2
168.75	191.25	S	1.5	3.0	2.2	1.1	0.4	0.0		0.0			8.2
191.25	213.75	SSW	1.6	3.3	3.3	2.7	1.1	0.3	0.0	0.0			12.2
213.75	236.25	SW	1.3	3.4	4.2	3.8	1.6	1.1	0.5	0.4	0.1	0.0	16.5
236.25	258.75	WSW	1.5	3.1	3.4	3.3	2.6	2.2	1.6	1.1	0.5	0.1	19.4
258.75	281.25	W	1.0	1.8	2.1	1.9	1.3	0.6	0.3	0.0	0.0		9.1
281.25	303.75	WNW	1.1	1.5	1.3	0.4	0.2	0.1	0.0	0.0			4.7
303.75	326.25	NW	0.9	1.0	0.6	0.3	0.0						2.8
326.25	348.75	NNW	1.0	1.1	0.5	0.1	0.1						2.7
348.75	11.25	N	0.8	0.8	0.5	0.1	0.0						2.2
Column Total (%)			18.0	28.7	21.9	14.5	7.7	4.3	2.5	1.6	0.7	0.2	

35-meters			Speed (cm/s)											
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	50 to 55	Row Total (%)
11.25	33.75	NNE	1.0	0.8	0.2	0.0								2.1
33.75	56.25	NE	0.7	0.5	0.3	0.0								1.6
56.25	78.75	ENE	0.8	0.8	0.3	0.0								2.0
78.75	101.25	E	1.0	1.1	0.3	0.0								2.4
101.25	123.75	ESE	1.4	1.6	0.8	0.2	0.0							3.9
123.75	146.25	SE	1.3	2.5	1.7	0.3	0.0							5.9
146.25	168.75	SSE	1.4	3.6	2.1	1.0	0.2	0.0						8.3
168.75	191.25	S	2.0	3.8	2.7	1.3	0.0	0.2						10.0
191.25	213.75	SSW	1.5	4.8	4.0	1.6	0.3	0.1						12.3
213.75	236.25	SW	1.4	4.0	3.4	1.4	0.7	0.5	0.3	0.2	0.1	0.0		12.0
236.25	258.75	WSW	1.3	2.9	2.6	1.4	1.4	1.6	1.3	1.1	0.8	0.5	0.1	15.1
258.75	281.25	W	1.4	2.3	1.8	1.0	1.7	1.3	1.1	0.7	0.2	0.2	0.0	11.8
281.25	303.75	WNW	1.4	1.3	1.1	0.6	0.5	0.4	0.2	0.0	0.0			5.7
303.75	326.25	NW	1.2	0.9	0.7	0.2	0.1	0.1						3.2
326.25	348.75	NNW	0.6	0.8	0.3	0.2	0.1							2.0
348.75	11.25	N	0.6	0.7	0.2	0.1		0.0						1.7
Column Total (%)			19.0	32.6	22.7	9.3	5.2	4.1	2.9	2.1	1.1	0.8	0.1	

Evaluation of these *quiescent-time* results indicated that the Venture Point farm site experiences relatively slow currents for approximately 50% of the time just above the seafloor (35-meter depth plane), while conditions near the surface (5-meter depth plane) suggests that flows are quite strong (>10 cm/s) for 65% of the time. Mid-water flows appear consistent with the lower water column, with a slight increase in the frequency of the quiescent periods with increasing depth. In general, the flows through this farm site, ignoring any vertical component that may develop as a result of bathymetric and/or infrastructure interferences, lack vertical stratification and can be regarded as laminar in nature.

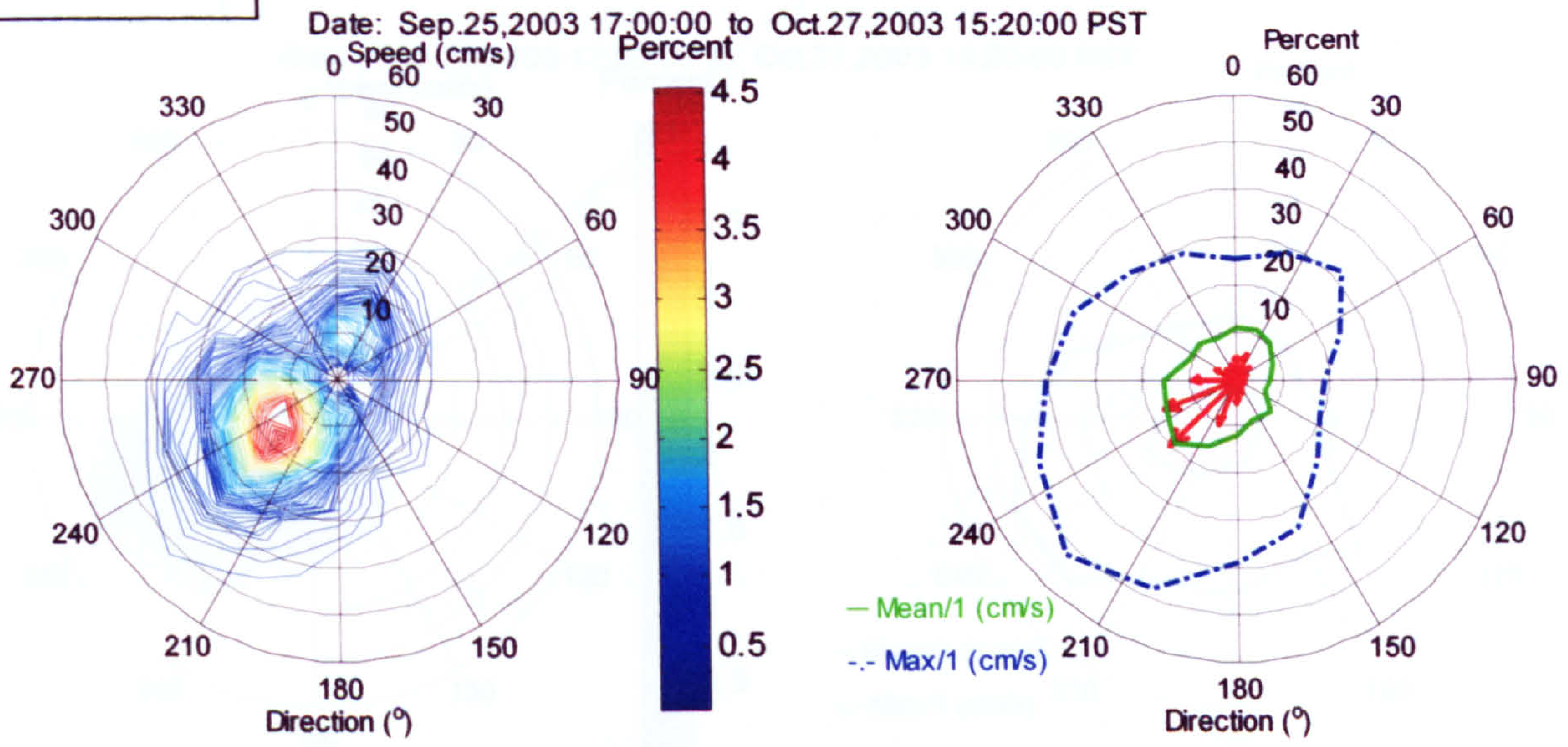
Figures 19 and 20 summarize the speed-direction tidal data graphically, using paired rosette plots for the upper depth planes (5 and 15 meter) and lower depth planes (25 and 35 meter), respectively. The right-hand-side plot shows the mean (green line) and maximum speed (dashed blue line) by direction, as well as the total percent occurrence (red vectors) in 22.5° segments. The left-hand-side illustrates the percent occurrence (frequency) of velocities by speed and direction.

The tidal data portrayed in Figures 19 and 20 reveal a predominant southwest flow at this farm site, regardless of depth. The majority of flow occurs within this compass quadrant (210° through 260° T), with an increasing westerly component (and decreasing velocity) occurring with depth.

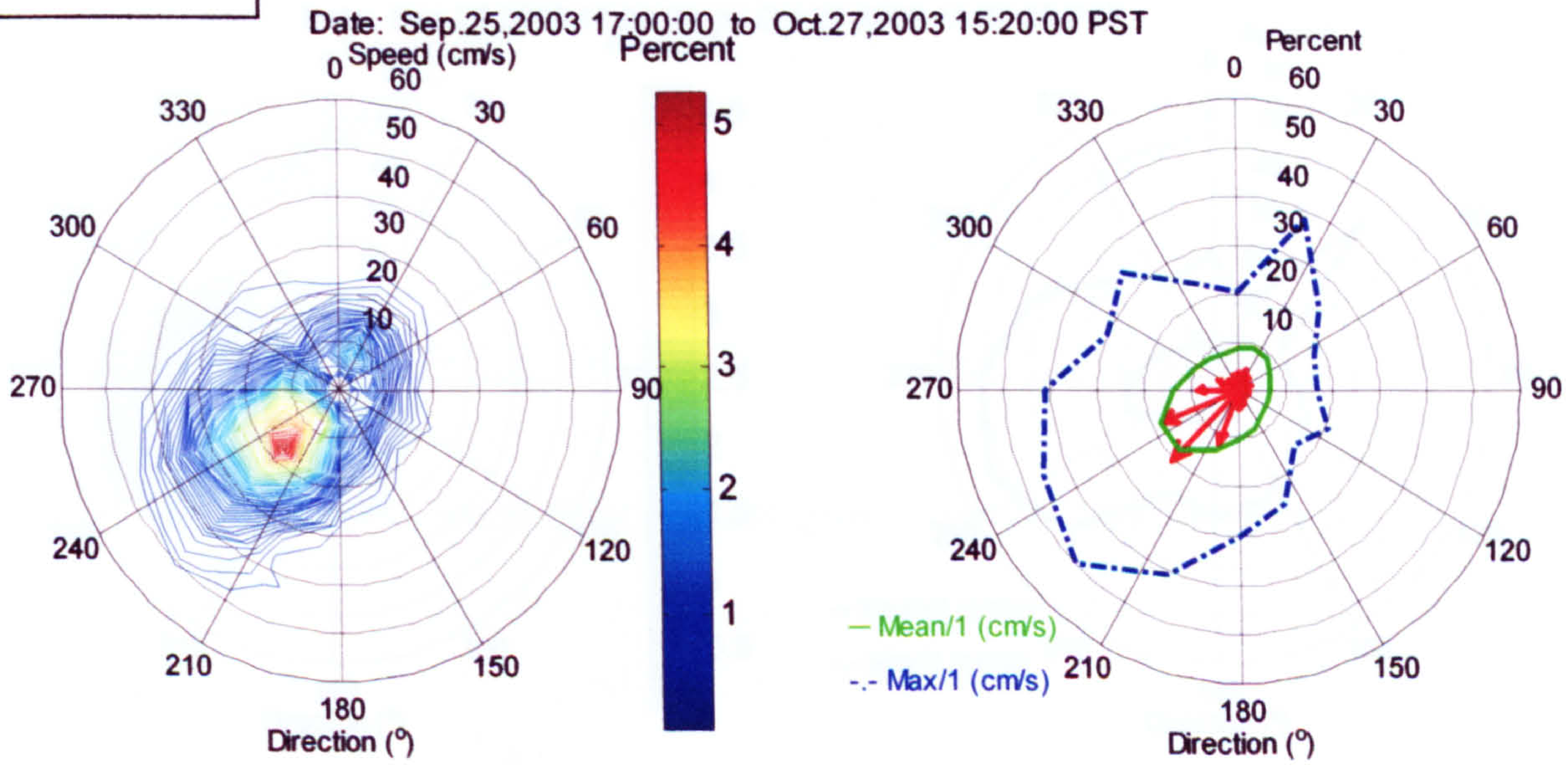
Figure 21 presents these tidal data in the form of a progressive Vector Diagram (PVD). The PVD for each of the depth planes at Venture Point show the cumulative speed and direction of flow over the entire 30-daye tidal cycle. The plotted positions on each PVD corresponds to horizontal displacements that would have occurred over a single day had the flow at the mooring (and at each depth) been the same as at the displaced position. These diagrams confirm a net movement, or residual flow, to the south-southwest at all depths measured, and validates the deployment position of the experimental longline system used for the shellfish component of this study.

Figure 19: Venture Point 600-khz ADCP tidal data acquired 25 September 2003 through 27 October 2003. Rosette summary shows directional frequency and velocities within 5 meter and 15 meter depth bins.

5 meters



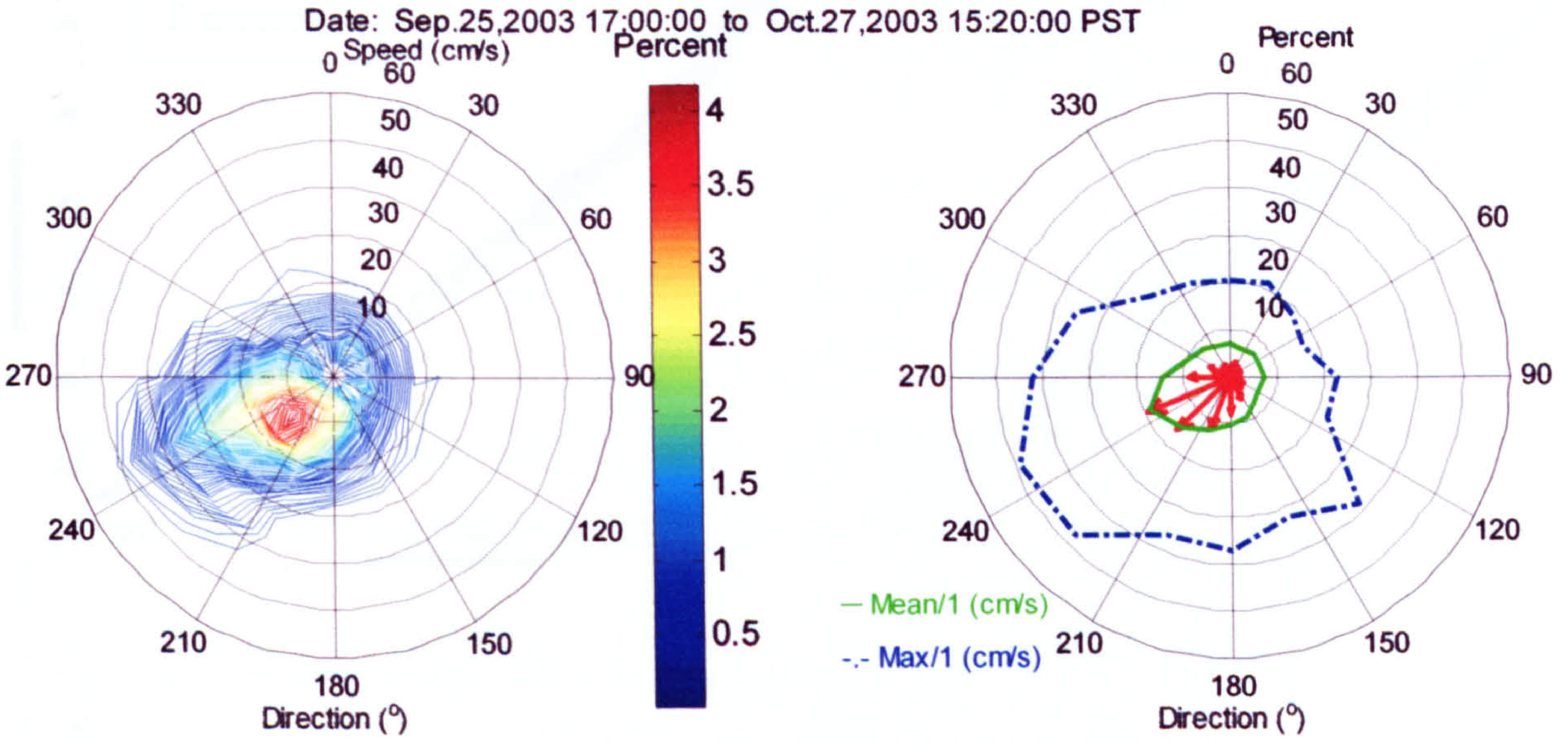
15 meters



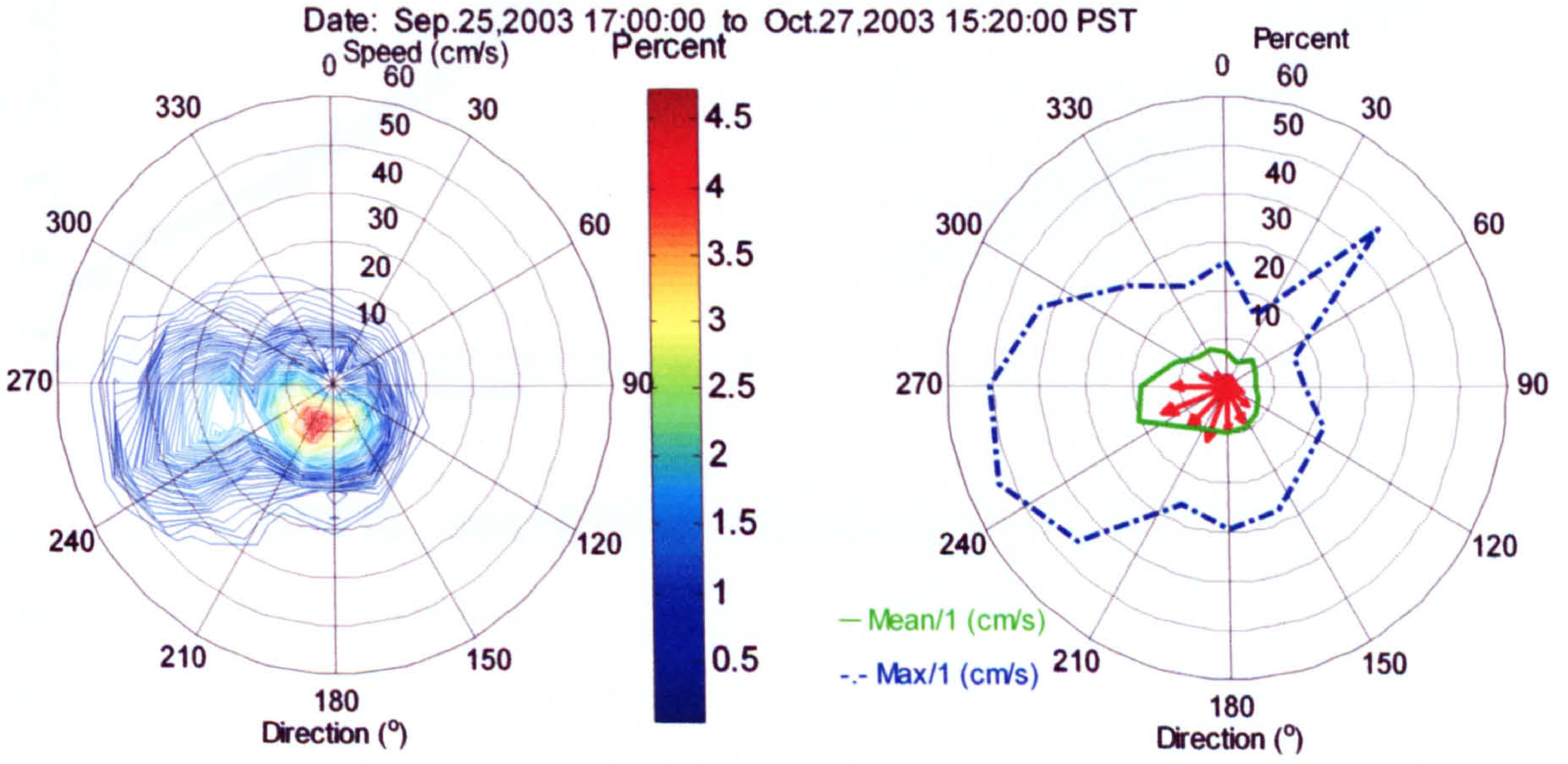
Experiment 25m Site : Venture Point Instrument AquaDopp Profiler

Figure 20: Venture Point 600-khz ADCP tidal data acquired 25 September 2003 through 27 October 2003. Rosette summary shows directional frequency and velocities within 25 meter and 35 meter depth bins.

25 meters

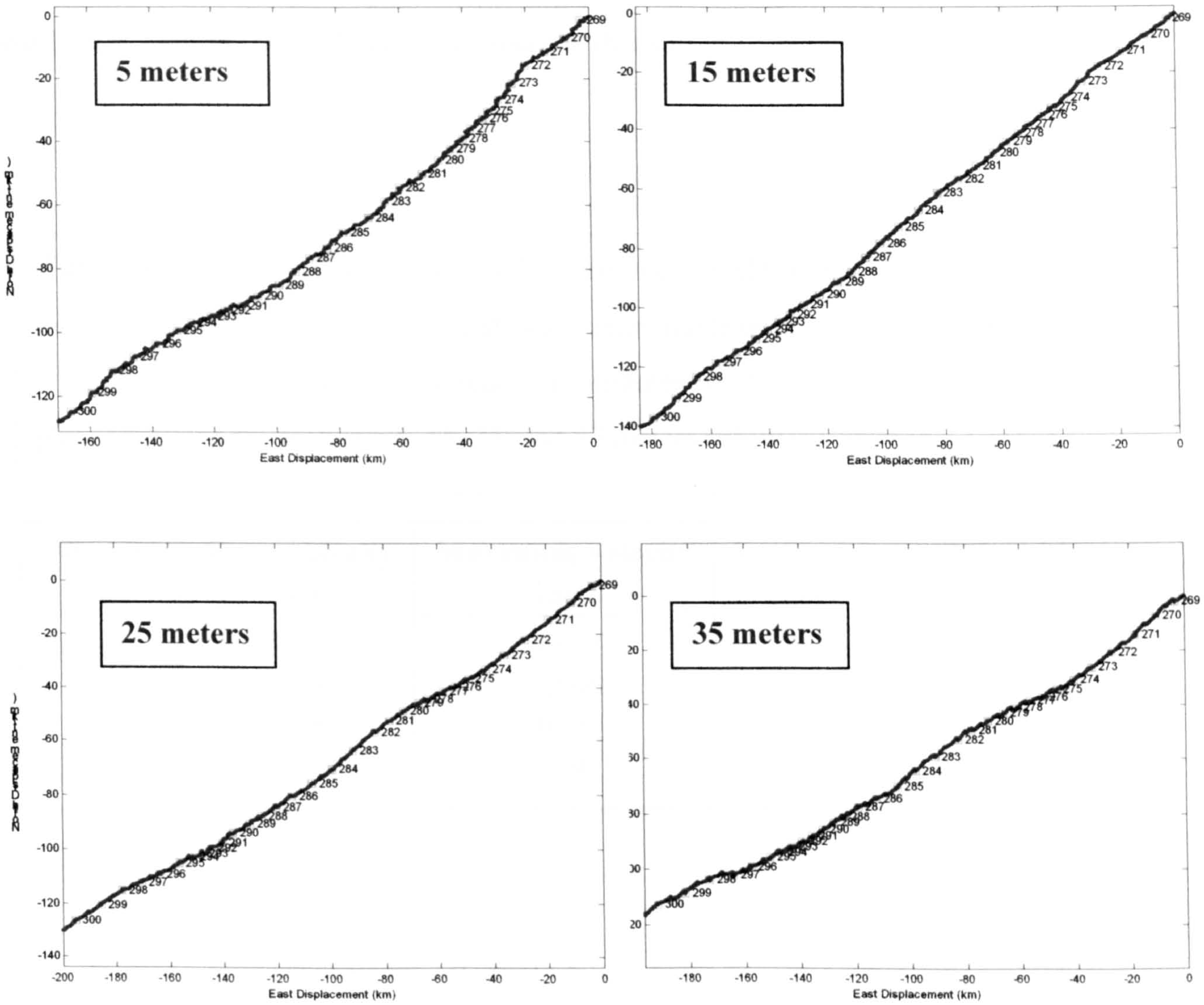


35 meters



Experiment: 05m Site : Venture Point Instrument: AquaDopp Profiler

Figure 21: Venture Point 600-khz ADCP tidal data acquired 25 September 2003 through 27 October 2003. Progressive Vector Diagrams (PVD's) for 5, 15, 25 and 35 meter depth planes. Values along line are Julian Days.



NOTE: *entire tidal current speed-direction data set used in*

Young Passage Farm Site:

The tidal current speed and direction data for the Young Passage farm site were acquired at a sampling interval of 10 minutes using a bottom-mounted, 600-khz ADCP mooring. The instrument was moored for a 35-day period (27 October 2003 through 1 December 2003), providing a total of 5,037 records. Table 6 presents the summary statistics for each depth plane (5, 15, 25 and 35 meters) assessed from the vertical profile data acquired through the deployment at this site.

Table 6: Summary statistics for tidal current meter (ADCP 600 khz RDI Broadband) deployed at the Young Passage farm site from 27 October through 1 December 2003. Sampling interval of 10 minutes; n= 5,037. Four depth planes selected for analysis: 5, 15, 25, and 35 meters below sea level (MLLW).

Depth Plane	**Mean Velocity** *cm/s*	**Maximum Velocity** *cm/s*	**Vector Averaged Speed** *cm/s*	*@ deg. T*
5 meters	9.87	50.07	2.36	158.00
15 meters	7.58	32.66	1.22	239.30
25 meters	7.55	40.29	1.74	284.30
35 meters	7.17	41.92	0.29	187.20

Mean tidal velocities reported within the water column at the Young Passage farm site were all below 10 cm/s, with the highest speed reported for the surface waters (5 meter depth) at 9.87 cm/s. Mid to lower water column velocities were all similar, with an overall average of speed of 7.43 cm/s. The vector-averaged tidal current speeds suggested highly variable current direction, with values ranging from 2.36 cm/s near the surface to 0.29 cm/s in the bottom waters. The direction of net flow varied from a southeasterly direction in the surface waters, to a northwesterly flow at 25 meters, and a southerly flow near the bottom. Peak velocities approached a full knot in the near-surface portion of the water column, with reduced maximums in the subsurface waters. Speed-direction frequency tables for the upper water column (5- and 15-meter depth planes) are presented in Table 7, and that for the lower portion of the water column (25- and 35-meter depth planes) in Table 8. Although there is some evidence of a bimodal tidal signal in these data, suggested by the green line on each frequency table, the strength of these tidal flows were limited. *Quiescent time* analysis of these data indicate that while surface waters comprise flows of <10 cm/s 60% of the time, the frequency of tidal velocities below this threshold occurred between 75 and 80% at depths at and below 15 meters.

Figures 22 and 23 present the tidal current information in graphical form, using the paired rosette plots to summarize frequency distribution of current speed-direction as well as the distribution of mean and maximum tidal flows for the upper depth planes (5- and 154-meter) and lower depth planes (25- and 35 meter), respectively. The 35-meter current data suggest a virtual stagnant tidal condition, with weak speeds equally distributed around the compass rosette, with no unimodal or bimodal distribution evident (Figure 23). At the surface there is an indication of a net flow towards the south, and this becomes more westerly within the 15-meter and 25-meter depth plane data.

The residual tidal flows, shown with Progressive Vector Diagrams (PVD's) in Figure 24, support the above observations. While the bottom currents reveal a confused cumulative flow over the 35-day tidal cycle, the surface (5-meter) through 25-meter depth planes reveal an increasing westerly component to the residual flows, particularly within the two mid-water planes. There was some notable confusion in the near-surface depth data, in terms of direction, and it is speculated that this flow pattern may be related to interference from the adjacent net-cage system.

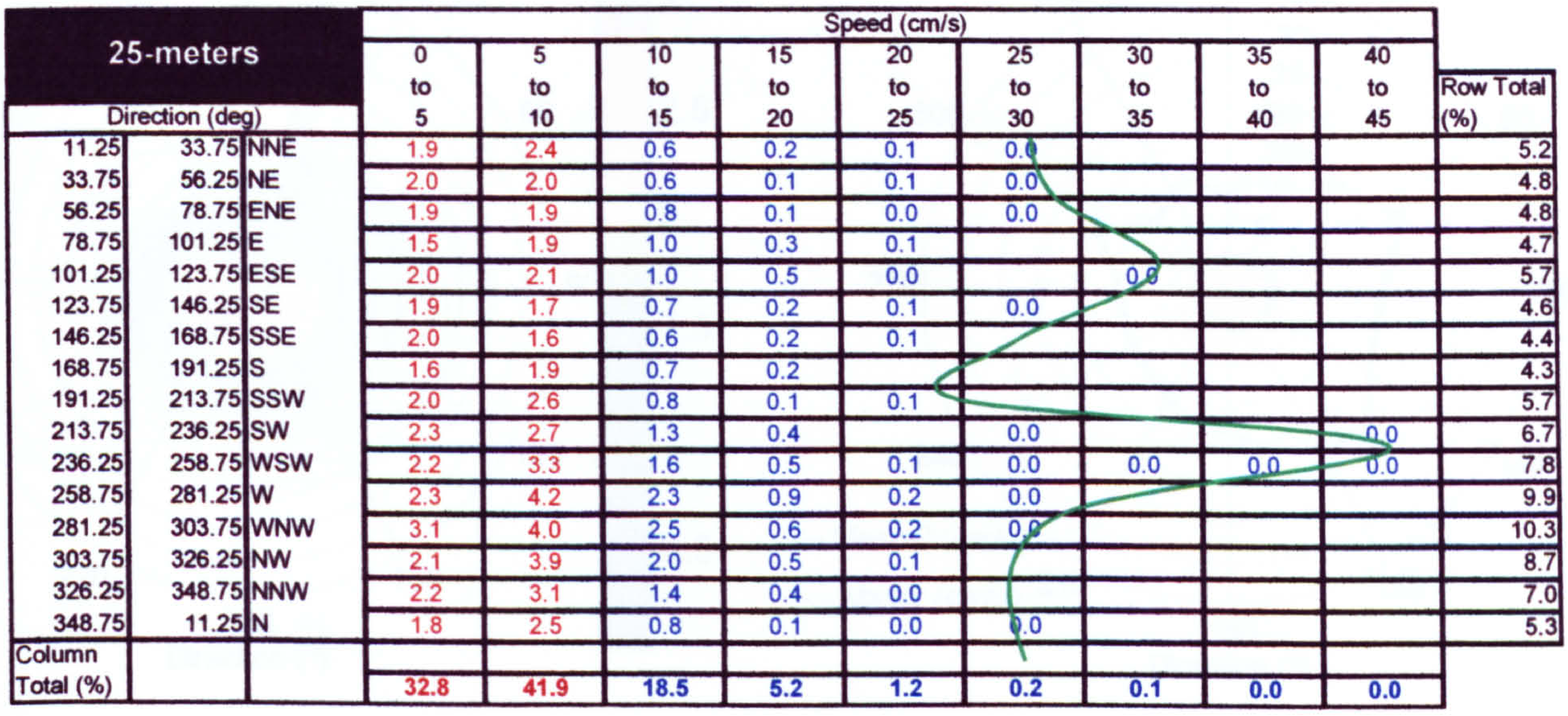
Despite the weak tidal flows at the Young Passage farm site, a measurable residual flow to the west-northwest validates the position of the experimental longline system deployed for the shellfish component of this research.

Table 7: Young Passage 600-khz ADCP tidal data acquired 27 October 2003 through 01 December 2003. Speed-direction frequency table for 5 meter and 15 meter depth bins. Directional trend over deployment cycle indicated with green line. Blue values suggest speeds at which potential suspension and re-suspension processes may occur. The red values area considered to represent periods within which depositional processes may occur (*quiet time*).

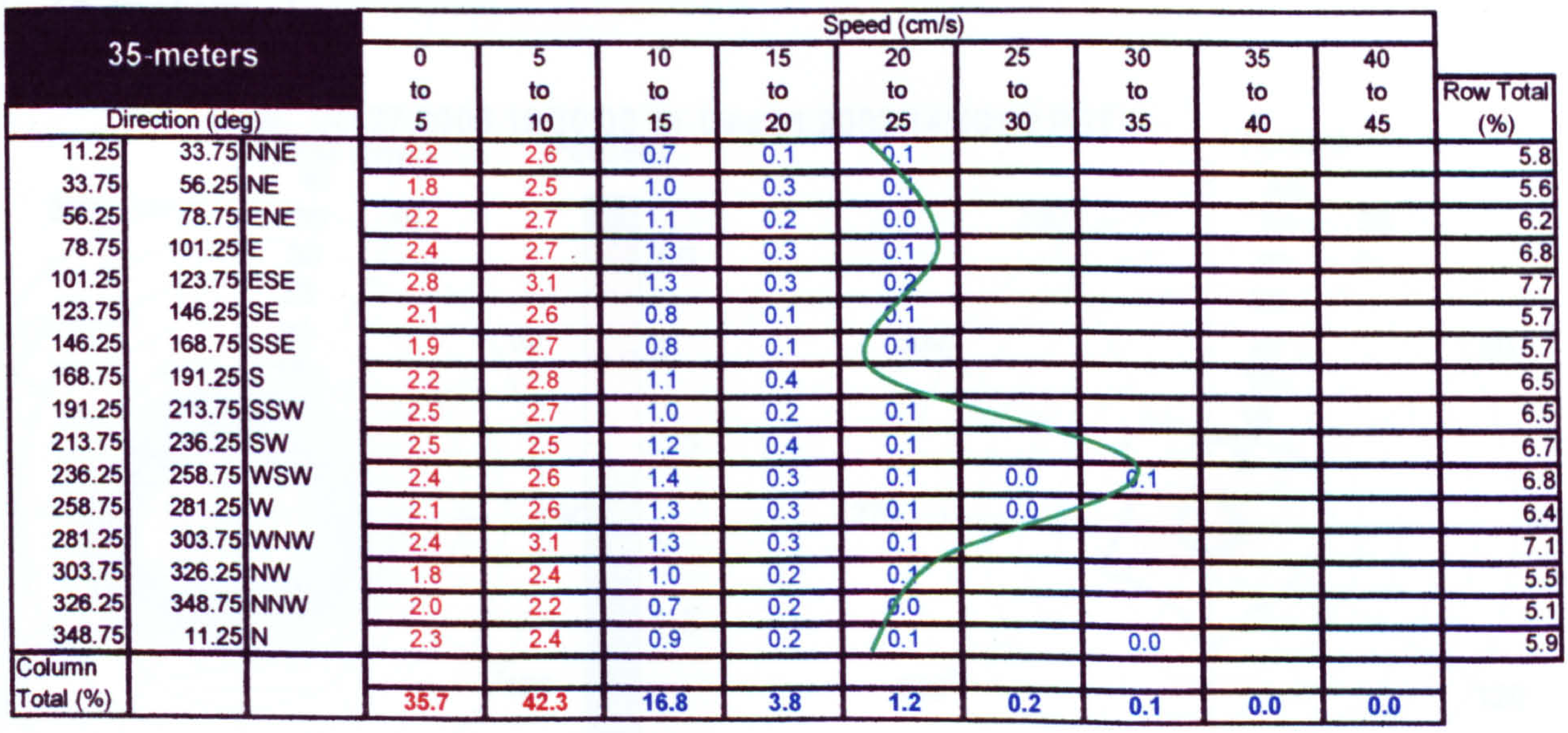
5-meters			Speed (cm/s)											
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	50 to 55	Row Total (%)
11.25	33.75	NNE	1.4	1.3	0.7	0.3	0.1	0.0						4.0
33.75	56.25	NE	1.4	1.9	0.9	0.4	0.4	0.1	0.1					5.1
56.25	78.75	ENE	1.2	1.6	1.1	0.6	0.4	0.1	0.1	0.1				5.2
78.75	101.25	E	1.4	2.3	1.7	1.0	0.6	0.3	0.1	0.1	0.1			7.6
101.25	123.75	ESE	1.5	2.6	1.9	1.0	0.6	0.2	0.2	0.0				8.0
123.75	146.25	SE	1.5	2.5	2.0	0.9	0.4	0.3	0.2	0.0				7.9
146.25	168.75	SSE	1.6	2.7	1.7	1.1	0.4	0.1	0.1					7.6
168.75	191.25	S	1.5	2.8	1.6	1.0	0.3	0.1	0.0					7.5
191.25	213.75	SSW	2.0	3.0	1.9	0.9	0.3	0.0	0.1					8.1
213.75	236.25	SW	2.2	3.2	1.7	0.8	0.2	0.1						8.3
236.25	258.75	WSW	1.6	3.2	1.9	0.9	0.4	0.0						7.9
258.75	281.25	W	1.3	2.8	1.6	0.6	0.0	0.0						6.4
281.25	303.75	WNW	1.7	2.0	1.4	0.4	0.1	0.1	0.0	0.0				5.7
303.75	326.25	NW	1.1	1.6	0.8	0.1	0.1	0.1						3.8
326.25	348.75	NNW	1.2	1.3	0.6	0.3	0.1		0.0					3.5
348.75	11.25	N	1.0	1.5	0.6	0.2	0.2	0.0						3.5
Column Total (%)			23.7	36.2	22.2	10.4	4.4	1.6	1.0	0.3	0.1	0.0	0.0	

15-meters			Speed (cm/s)										
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	45 to 50	Row Total (%)
11.25	33.75	NNE	2.4	1.8	0.7	0.1	0.1						5.1
33.75	56.25	NE	1.7	1.7	0.6	0.0		0.0					4.1
56.25	78.75	ENE	1.9	2.3	0.8	0.2	0.0		0.0				5.3
78.75	101.25	E	1.8	2.2	1.1	0.4	0.1	0.1	0.0				5.6
101.25	123.75	ESE	2.1	2.9	1.4	0.4	0.1	0.1					7.0
123.75	146.25	SE	1.8	2.5	1.4	0.2	0.1	0.0					6.0
146.25	168.75	SSE	1.6	2.2	0.6	0.2	0.1						4.7
168.75	191.25	S	1.8	2.2	0.8	0.2	0.1						5.1
191.25	213.75	SSW	1.8	2.9	1.3	0.3	0.1						6.5
213.75	236.25	SW	1.7	3.8	1.9	0.5	0.1						7.9
236.25	258.75	WSW	2.2	3.6	2.7	0.9	0.0	0.0	0.0				9.5
258.75	281.25	W	2.4	3.9	2.2	0.6	0.1						9.2
281.25	303.75	WNW	2.3	3.6	1.5	0.3	0.1						7.8
303.75	326.25	NW	1.8	2.9	0.8	0.2		0.0	0.0				5.7
326.25	348.75	NNW	1.7	2.6	0.8	0.1	0.1	0.0					5.3
348.75	11.25	N	2.1	2.5	0.6	0.1	0.0						5.3
Column Total (%)			31.1	43.7	19.3	4.5	1.1	0.3	0.1	0.0	0.0	0.0	

Table 8: Young Passage 600-khz ADCP tidal data acquired 27 October 2003 through 01 December 2003. Speed-direction frequency table for 25 meter and 35 meter depth bins. Directional trend over deployment cycle indicated with green line. Blue values suggest speeds at which potential suspension and re-suspension processes may occur. The red values area considered to represent periods within which depositional processes may occur (*quiet time*).



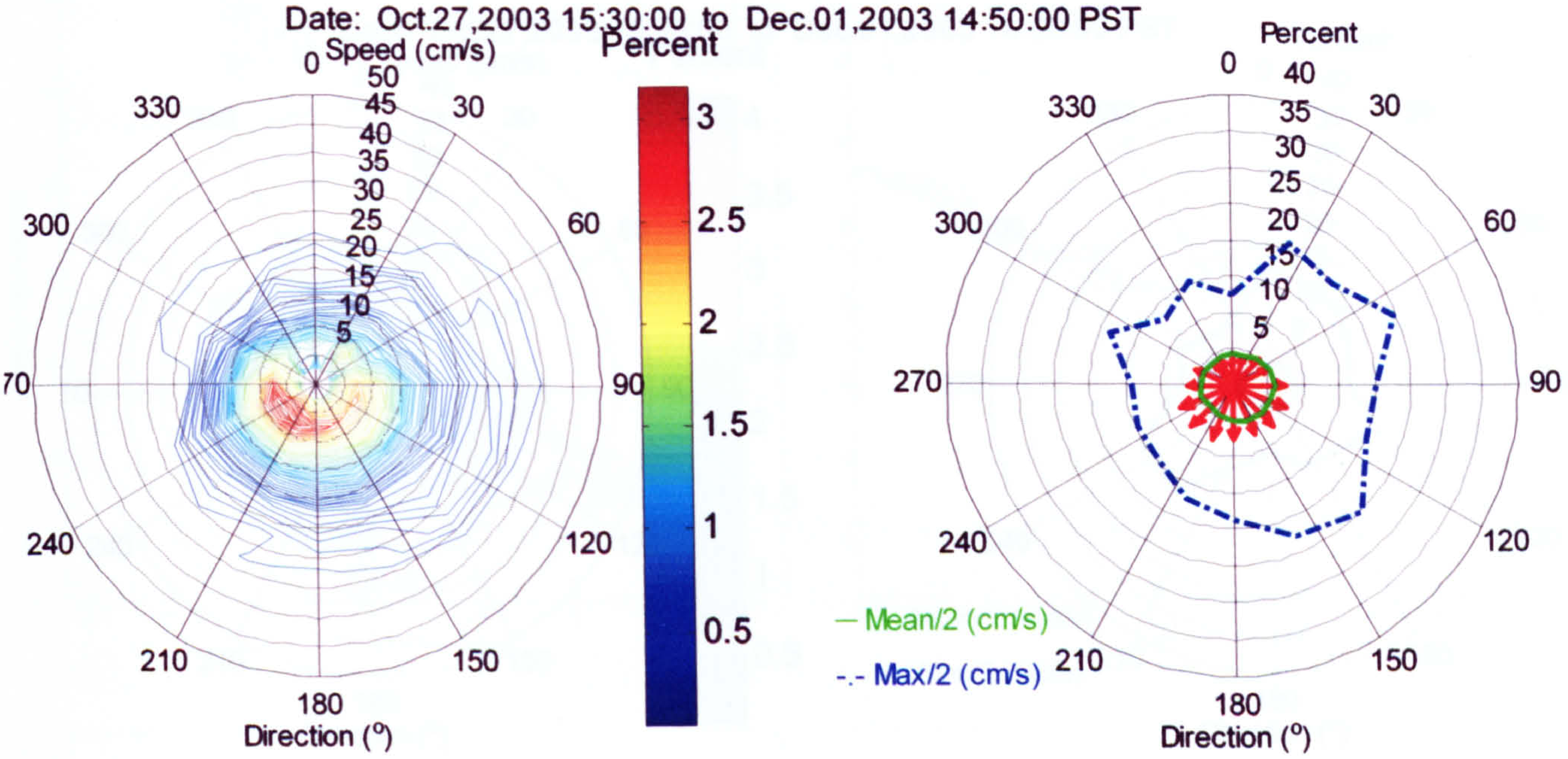
25-meters			Speed (cm/s)									
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	Row Total (%)
11.25	33.75	NNE	1.9	2.4	0.6	0.2	0.1	0.0				5.2
33.75	56.25	NE	2.0	2.0	0.6	0.1	0.1	0.0				4.8
56.25	78.75	ENE	1.9	1.9	0.8	0.1	0.0	0.0				4.8
78.75	101.25	E	1.5	1.9	1.0	0.3	0.1					4.7
101.25	123.75	ESE	2.0	2.1	1.0	0.5	0.0		0.0			5.7
123.75	146.25	SE	1.9	1.7	0.7	0.2	0.1	0.0				4.6
146.25	168.75	SSE	2.0	1.6	0.6	0.2	0.1					4.4
168.75	191.25	S	1.6	1.9	0.7	0.2						4.3
191.25	213.75	SSW	2.0	2.6	0.8	0.1	0.1					5.7
213.75	236.25	SW	2.3	2.7	1.3	0.4		0.0			0.0	6.7
236.25	258.75	WSW	2.2	3.3	1.6	0.5	0.1	0.0	0.0	0.0	0.0	7.8
258.75	281.25	W	2.3	4.2	2.3	0.9	0.2	0.0				9.9
281.25	303.75	WNW	3.1	4.0	2.5	0.6	0.2	0.0				10.3
303.75	326.25	NW	2.1	3.9	2.0	0.5	0.1					8.7
326.25	348.75	NNW	2.2	3.1	1.4	0.4	0.0					7.0
348.75	11.25	N	1.8	2.5	0.8	0.1	0.0	0.0				5.3
Column Total (%)			**32.8**	**41.9**	**18.5**	**5.2**	**1.2**	**0.2**	**0.1**	**0.0**	**0.0**	



35-meters			Speed (cm/s)									
Direction (deg)			0 to 5	5 to 10	10 to 15	15 to 20	20 to 25	25 to 30	30 to 35	35 to 40	40 to 45	Row Total (%)
11.25	33.75	NNE	2.2	2.6	0.7	0.1	0.1					5.8
33.75	56.25	NE	1.8	2.5	1.0	0.3	0.1					5.6
56.25	78.75	ENE	2.2	2.7	1.1	0.2	0.0					6.2
78.75	101.25	E	2.4	2.7	1.3	0.3	0.1					6.8
101.25	123.75	ESE	2.8	3.1	1.3	0.3	0.2					7.7
123.75	146.25	SE	2.1	2.6	0.8	0.1	0.1					5.7
146.25	168.75	SSE	1.9	2.7	0.8	0.1	0.1					5.7
168.75	191.25	S	2.2	2.8	1.1	0.4						6.5
191.25	213.75	SSW	2.5	2.7	1.0	0.2	0.1					6.5
213.75	236.25	SW	2.5	2.5	1.2	0.4	0.1					6.7
236.25	258.75	WSW	2.4	2.6	1.4	0.3	0.1	0.0	0.1			6.8
258.75	281.25	W	2.1	2.6	1.3	0.3	0.1	0.0				6.4
281.25	303.75	WNW	2.4	3.1	1.3	0.3	0.1					7.1
303.75	326.25	NW	1.8	2.4	1.0	0.2	0.1					5.5
326.25	348.75	NNW	2.0	2.2	0.7	0.2	0.0					5.1
348.75	11.25	N	2.3	2.4	0.9	0.2	0.1		0.0			5.9
Column Total (%)			**35.7**	**42.3**	**16.8**	**3.8**	**1.2**	**0.2**	**0.1**	**0.0**	**0.0**	

Figure 22: Young Passage 600-khz ADCP tidal data acquired 27 October 2003 through 01 December 2003. Rosette summary shows directional frequency and velocities within 5 meter and 15 meter depth bins.

5 meters



15 meters

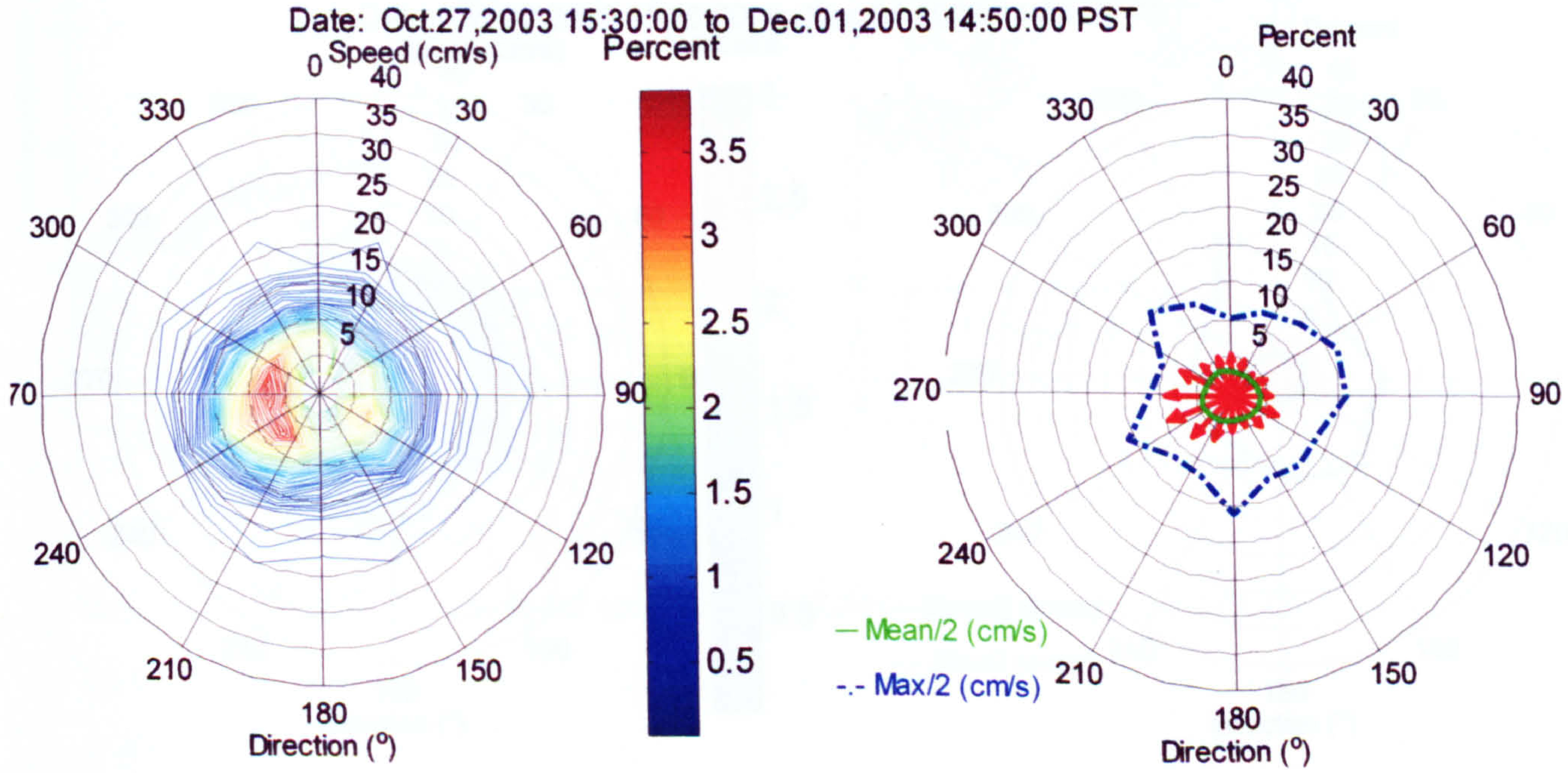
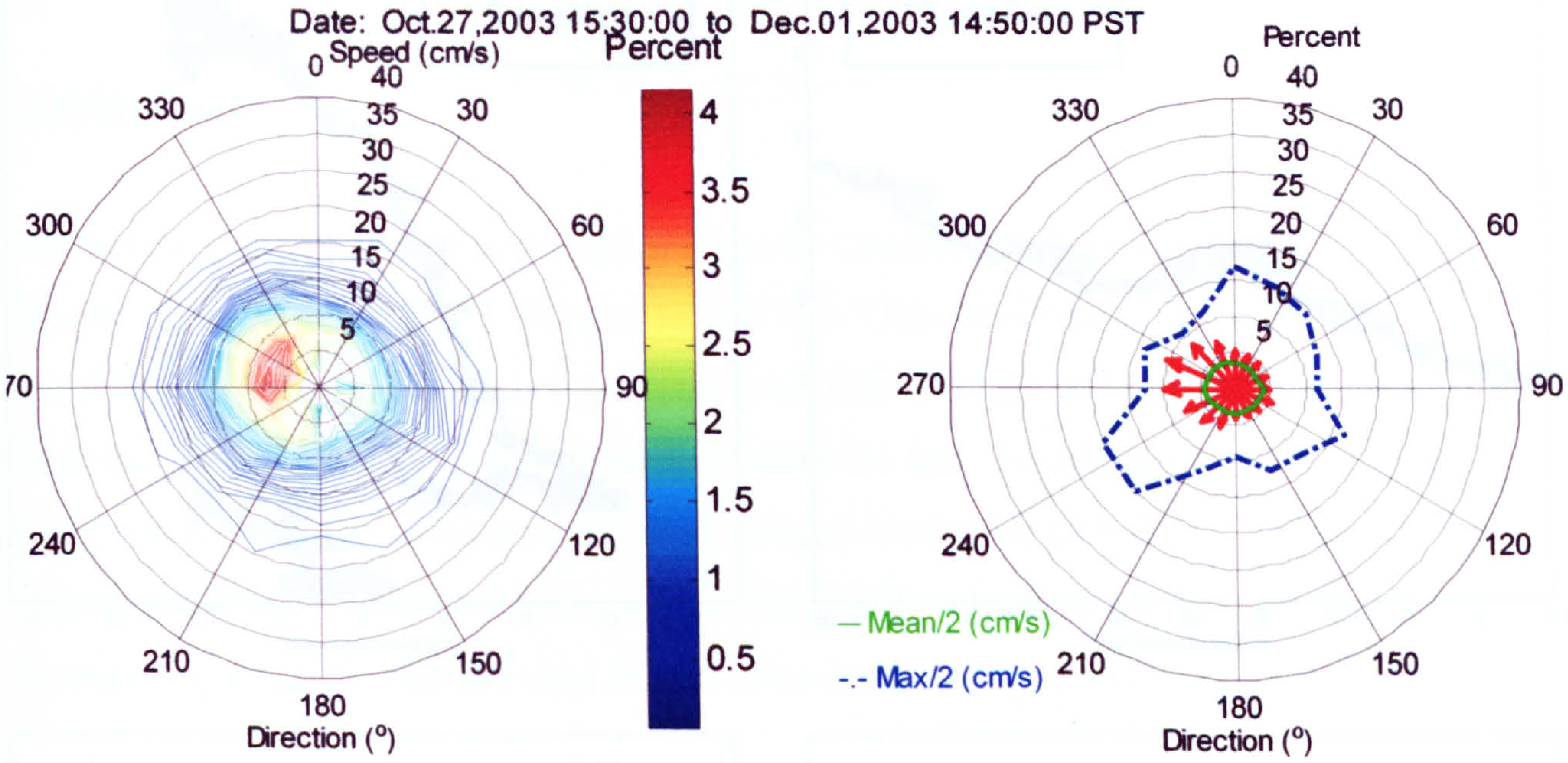


Figure 23: Young Passage 600-khz ADCP tidal data acquired 27 October 2003 through 01 December 2003. Rosette summary shows directional frequency and velocities within 25 meter and 35 meter depth bins.

25 meters



35 meters

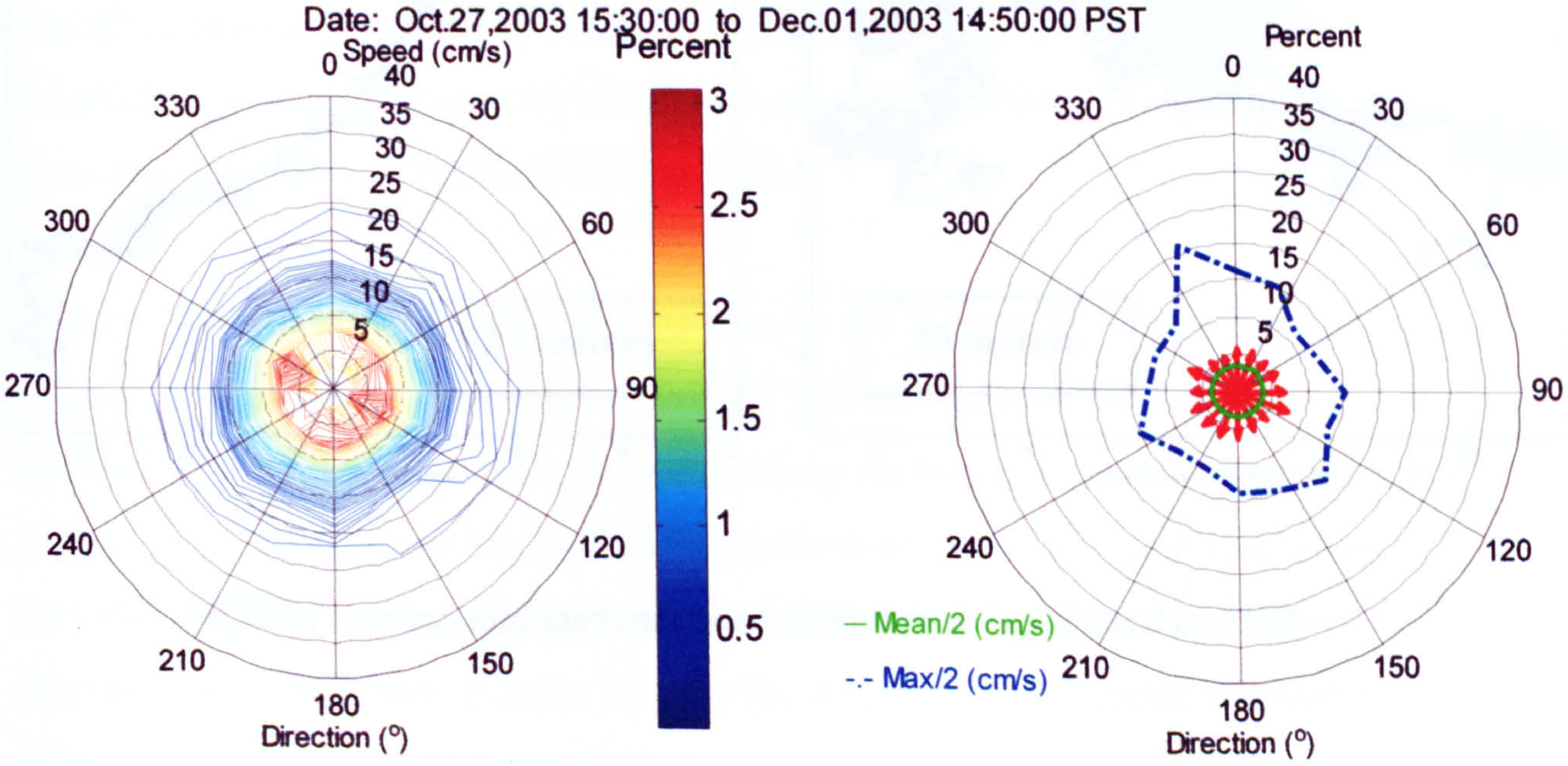
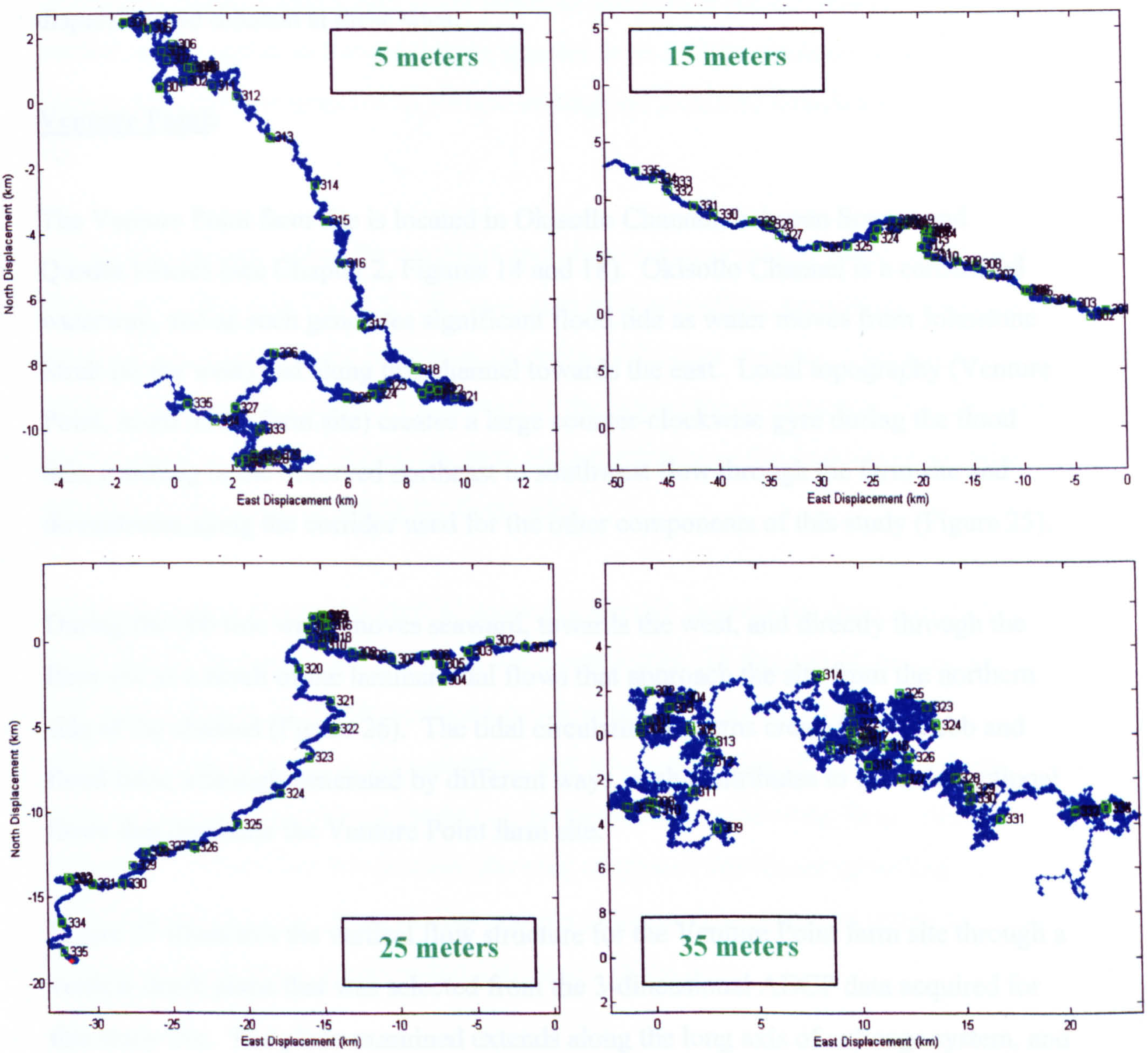


Figure 24: Young Passage 600-khz ADCP tidal data acquired 28 October 2003 through 01 December 2003. Progressive Vector Diagrams (PVD's) for 5, 15, 25 and 35 meter depth planes. Values along each blue line are Julian Days.



NOTE: *entire tidal current speed-direction data set used in PVD*

3.4.2 Tidal Circulation

The horizontal component of the tidal circulation data, acquired from the ADCP surveys, were compiled in a pair of summary plots that illustrate the generalized pattern of water flows that occur in the vicinity of each of the study farm sites. A second figure is provided to illustrate the vertical component of these flows, indicating the degree to which upwelling and down-welling processes may influence waste material dispersion and dilution at these sites.

Venture Point:

The Venture Point farm site is located in Okisollo Channel, between Sonora and Quadra Islands (see Chapter 2, Figures 14 and 18). Okisollo Channel is a constricted waterway, and as such generates significant flood tide as water moves from Johnstone Strait (to the west) and along this channel towards the east. Local topography (Venture Point, south of the farm site) creates a large counter-clockwise gyre during the flood tide, resulting in the observed northeast to southwest flow through the farm site and downstream along the corridor used for the other components of this study (Figure 25).

During the ebb tide water moves seaward, towards the west, and directly through the farm site as a result of the laminar tidal flows that approach the site from the northern side of the channel (Figure 26). The tidal circulation patterns created by the ebb and flood tides, although generated by different ways, each contributes to the unidirectional flows that dominate the Venture Point farm site.

Figure 27 illustrates the vertical flow structure for the Venture Point farm site through a vertical depth plane that was selected from the 3-dinensional ADCP data acquired for this study site. The plane examined extends along the long axis of net-cage system, and 240 meters in the upstream and 240 meters in the downstream direction (red dotted line, Figure 26). The plane comprises a cross-sectional area defined by the depth (approximately 60 meters; 2-meter bins) by the length of the horizontal component (700 meters; 20-meter data separation).

The vertical velocity data (mm/s) summarized for this plane (Figure 27) shows the pattern of downward flows (gradient of red contours; negative flows) and upwelling, or positive flows (blue-shaded contours). The unidirectional nature of flow at this site results in a left to right movement of water across this depth plane regardless of tidal state (discussed previously). Despite the strong horizontal flows through this site, the vertical flow component shows minimal structure, with negligible upward or downward flows. In general, the water column suggests a rather laminar flow with vertical flows of near 0.0 mm/s or with a very slight negative component (to -0.5 mm/s). In the near-surface region (down to 5 meters) there appears to be a slight upwards flow (+1.5 to +3.0 mm/s), likely in response to surface mixing and localized turbulence (Figure 27, A).

As the tidal current strikes the northeast end of the net cage system water is deflected downwards, shown in Figure 27 (B) as an area of downwelling through the upper 20-25 meters of the water column (approximate depth of the cage system). This down-welling area extends outwards (upstream) of the system for an estimated 50 meters, and is speculated to 'pile up' behind the farm infrastructure over the duration of the tidal activity (in this example during the flood tide).

Although current flows could not be measured directly beneath the farm system, it is likely that flows remain laminar along the lower depths (25 through 60 meters) as indicated in Figure 27 for the upstream and downstream regions. It is further speculated that flows along the underside, and along either side of the farm system, cause considerable turbulence (wake) on the downstream side of the farm. Area D (Figure 27) exemplifies this process, showing an area of significant upwelling (velocities of +3.5 to +5.0 mm/s) that appears to disperse, weaken and return to background turbulence levels within 150 meters of the farm.

Figure 25: Averaged Flood Tide circulation pattern for upper water column (5-20 meters) at the Venture Point farm site. Data derived from 300-khz ADCP survey of farm site. Arrows indicate direction of flow, with their respective lengths providing a relative measure of current velocity. The position of farm system is shown in blue.

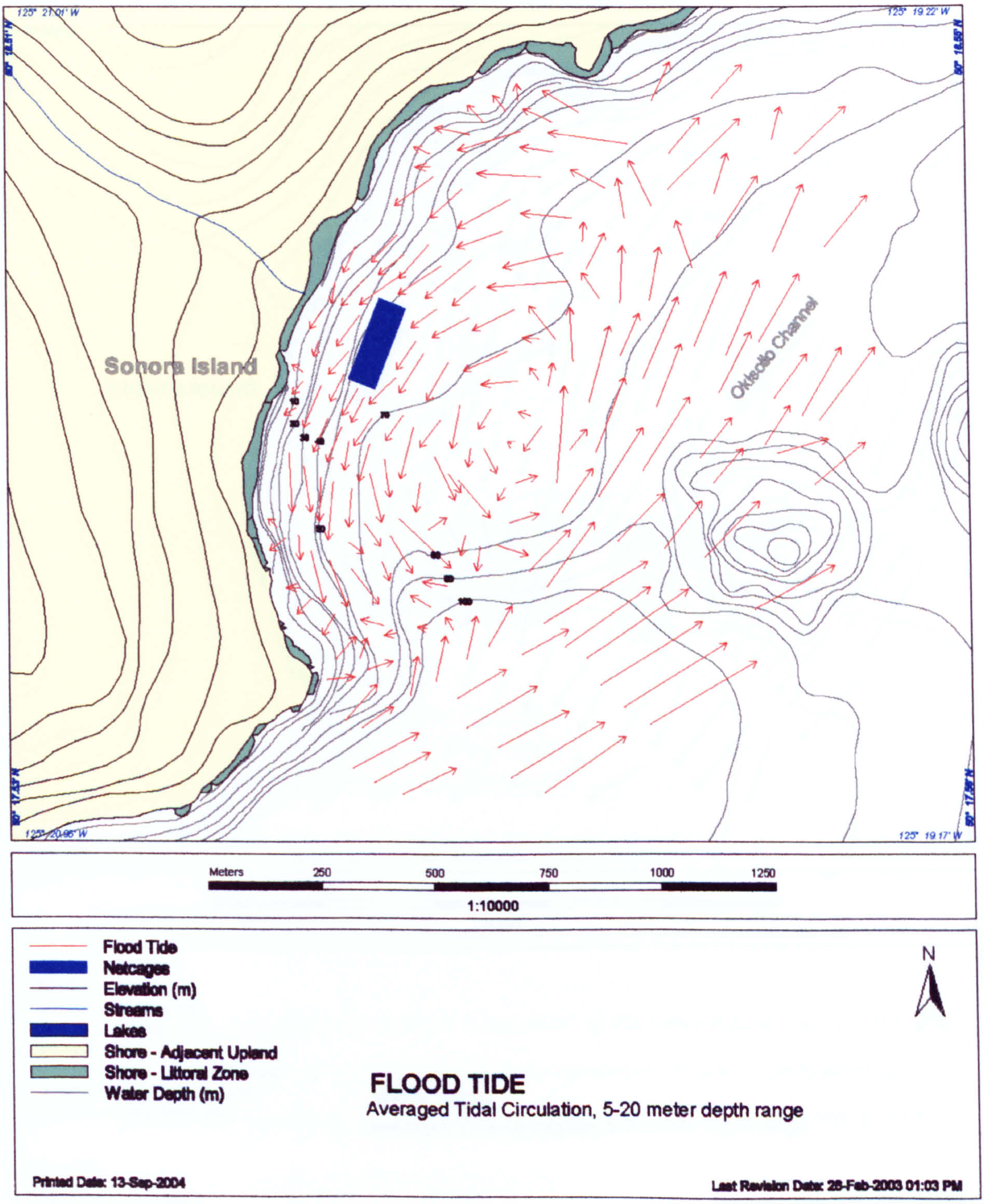


Figure 26: Averaged Ebb Tide circulation pattern for upper water column (5-20 meters) at the Venture Point farm site. Data derived from 300-khz ADCP survey of farm site. Arrows indicate direction of flow, with their respective lengths providing a relative measure of current velocity. The position of farm system is shown in blue. Centerline transect used to examine vertical component of the flow shown in red.

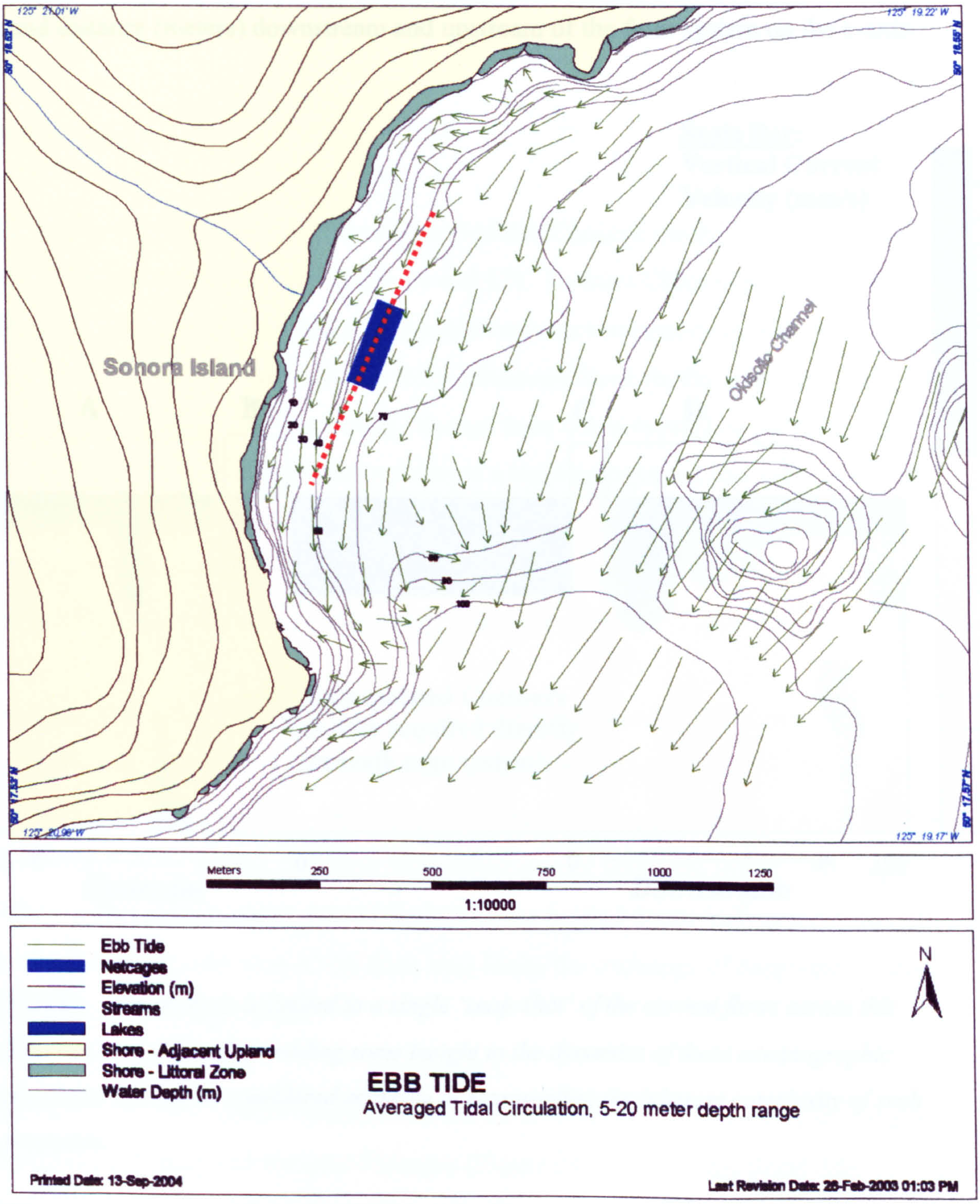
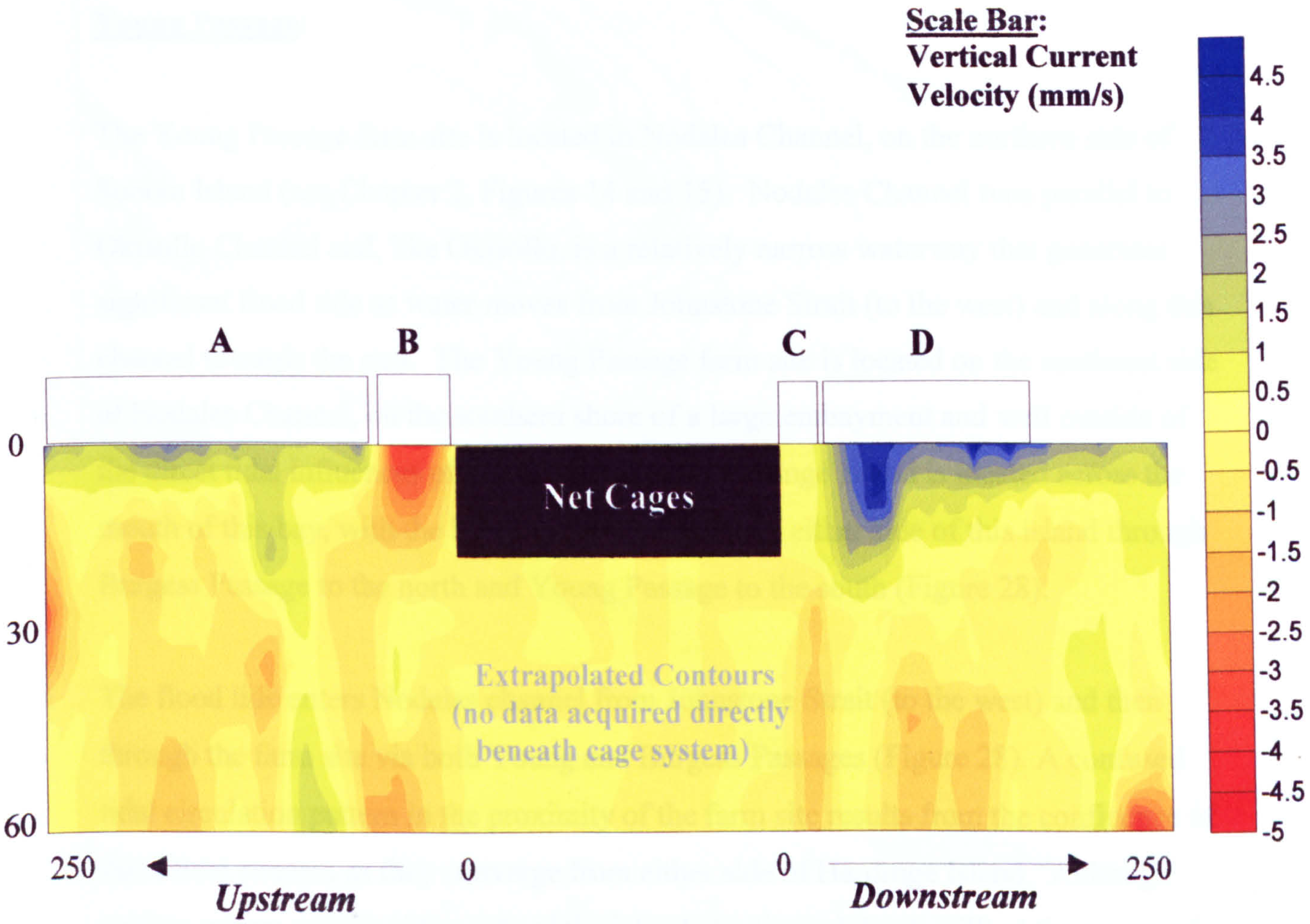


Figure 27: Vertical current velocities (mm/s) through a 700-meter cross-section of the Venture Point farm site. Data extracted from flood-tide ADCP record for the farm site, showing upwelling (blue) and down-welling (red) currents along a randomly selected centerline transect. Farm system shown by black shading; area beneath is extrapolated from data and is not considered accurate. Water depth (0-60 meters) on y-axis, and distance (meters) downstream and upstream of the farm system on the x-axis.



NOTE: *this analysis is limited to a single 'snap-shot' of the current flows across this farm site, and although providing some insight to the dynamics of these oceanographic processes, can not be considered accurate in representing the inherent complexity of such processes.*

A back-eddy effect is also caused by the farm, which results in an area (Figure 27, C) that is largely protected from the tidal flows. This area extends approximately 30-50 meters downstream and seems to indicate a region of no vertical flows. The horizontal component of the circulation (Figures 25, 26) suggests decreased velocities in this area, although spatial resolution of this analysis (20-meter) limits a detailed assessment of this effect.

Young Passage:

The Young Passage farm site is located in Nodales Channel, on the northern side of Sonora Island (see Chapter 2, Figures 14 and 15). Nodales Channel runs parallel to Okisollo Channel and, like Okisollo, is a relatively narrow waterway that generates significant flood tide as water moves from Johnstone Strait (to the west) and along this channel towards the east. The Young Passage farm site is located on the southeast side of Nodales Channel, on the southern shore of a large embayment and well outside of the direct tidal influences of the main channel. Hardinge Island is located across the mouth of this bay, with the inside waters accessed on either side of this island through Burgess Passage to the north and Young Passage to the south (Figure 28).

The flood tide enters Nodales channel from Johnstone Strait (to the west) and then through the farm site via both Young and Burgess Passages (Figure 28). A confused tidal circulation pattern in the proximity of the farm site results from the confluence of these tidal streams as they converge from either side of Hardinge Island. Although surface waters pass to either side of the island, the limited depth (sill) at the center of Young Passage (to the west of the farm site) limits the exchange of deepwater to the north side of Hardinge Island.

During the ebb tide surface water moves seaward, towards the west, and out of the bay through both Young and Burgess Passages (Figure 29). As with the flood tide, deepwater exchange with Nodales Channel will occur primarily through Burgess Passage to the north of Hardinge Island.

Figure 28: Averaged Flood Tide circulation pattern for upper water column (5-20 meters) at the Young Passage farm site. Data derived from 300-khz ADCP survey of farm site. Arrows indicate direction of flow, with their respective lengths providing a relative measure of current velocity. The position of farm system is shown in blue.

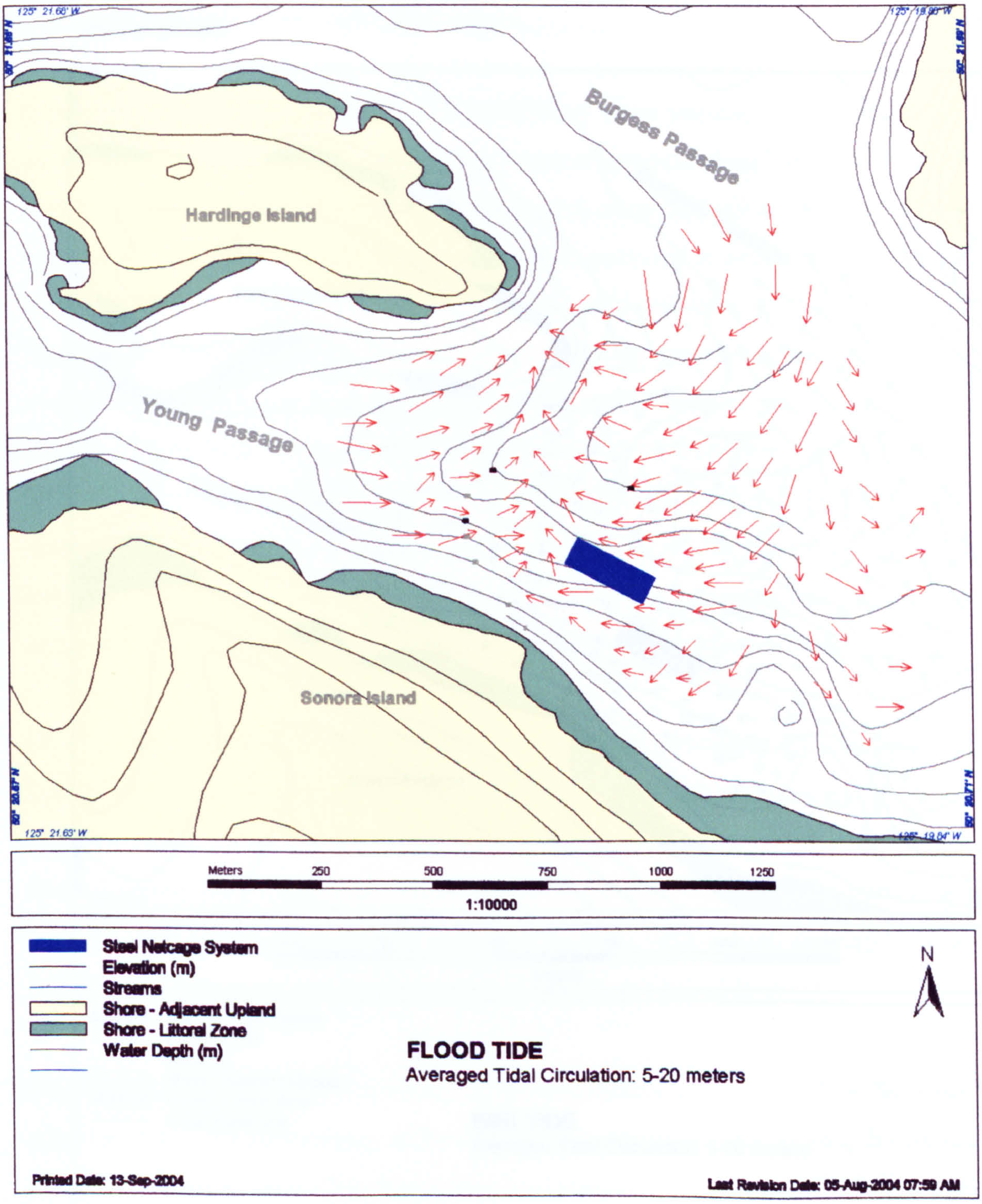
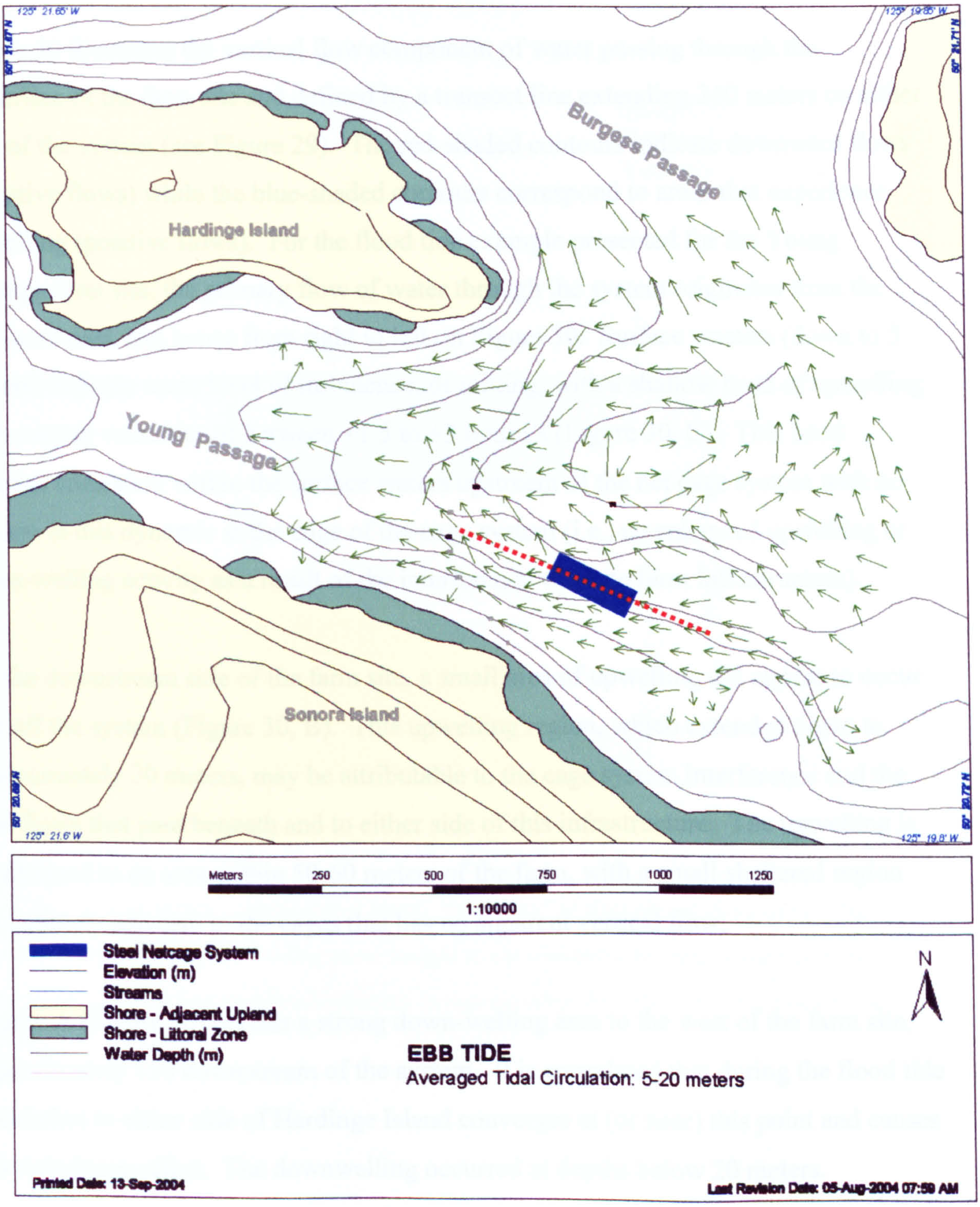


Figure 29: Averaged Ebb Tide circulation pattern for upper water column (5-20 meters) at the Young Passage farm site. Data derived from 300-khz ADCP survey of farm site. Arrows indicate direction of flow, with their respective lengths providing a relative measure of current velocity. The position of farm system is shown in blue. Centerline transect used to examine vertical component of the flow shown in red.



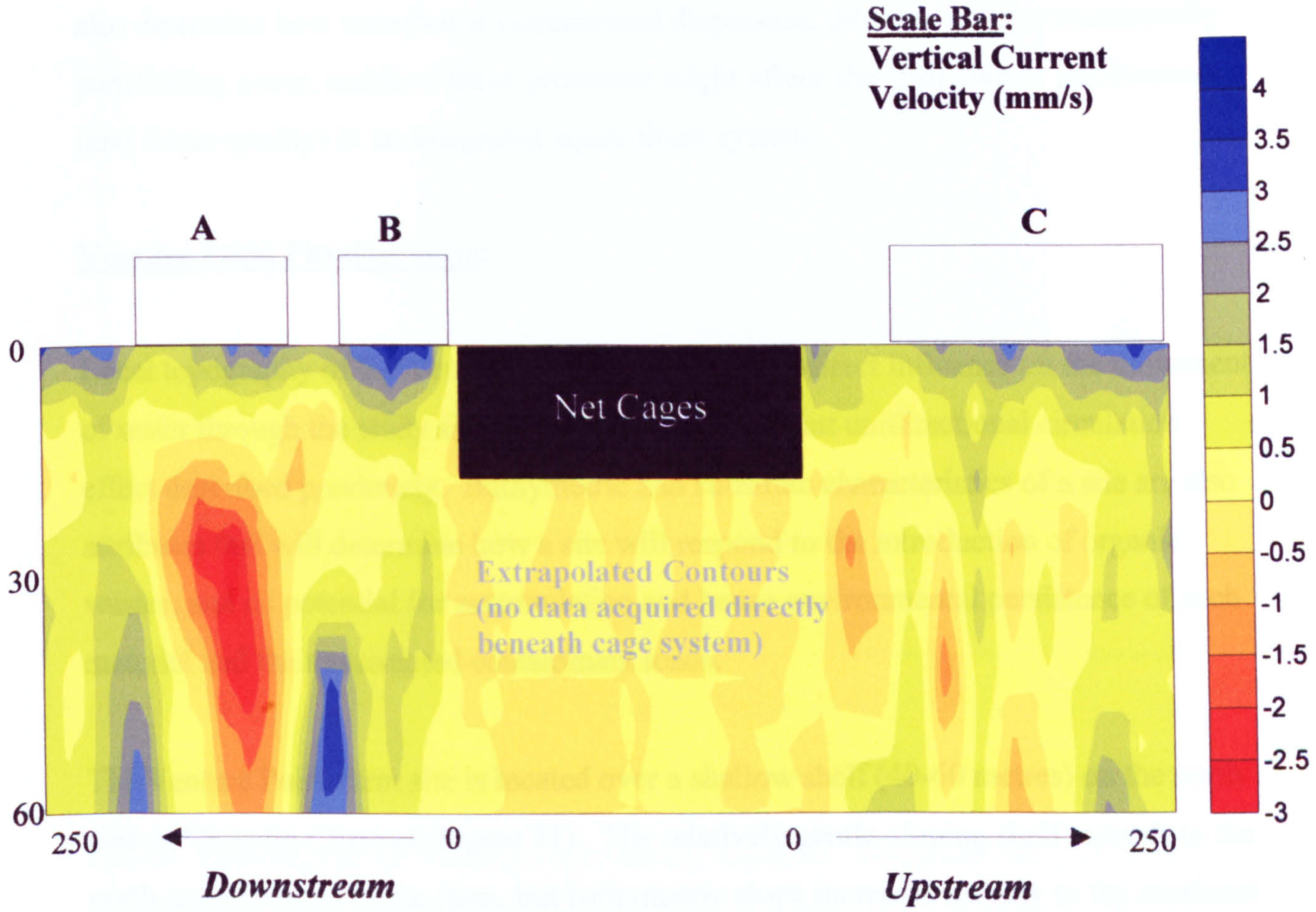
Despite these influential topographic and bathymetric features water circulation at the Young Passage farm site revealed a net westerly flood tide flow, albeit it with low velocities and a generally confused (non-laminar) pattern. Although tidal flows consisted of a distinct pattern in the surface waters (illustrated in Figure 29), the lower water column comprised a weak and variable current regime consistent with the results acquired through the fixed meter deployment.

Figure 30 illustrates the vertical flow component of water passing through the centerline of the farm site and defined by a transect line extending 240 meters on either side of the system (see Figure 29). The red-shaded contours indicate downward flows (negative flows) while the blue-shaded contours correspond to areas that experience upwelling (positive flows). For the flood tide example presented for the Young Passage farm site, the primary flow of water through the system originates from the east-northeast, and hence from right to left on Figure 30. Surface wasters (down to 5 meters) indicate some level of turbulence, or mixing, with a shallow band of upwelling with current velocities of between +1.5 to +3.0 mm/s (Figure 30, C). This band remains consistent within the surface waters upstream of the net cage system with no change in this dynamic at the edge of the cage system (i.e., no enhanced upwelling or down-welling activity as a result of the interference with the farm infrastructure).

On the downstream side of the farm site, a small area of upwelling did appear to occur just off the system (Figure 30, B). This upwelling region, which extended down to approximately 20 meters, may be attributable to the cage system interference and the tidal flows that pass beneath and to either side of this infrastructure. The upwelling is constrained to an area within 50-60 meters of the farm, with a small sheltered region immediately adjacent to the cages that has no apparent vertical flow.

Region A (Figure 30) reveals a strong down-welling area to the west of the farm site, approximately 150 downstream of the system. It is speculated that during the flood tide circulation to either side of Hardinge Island converges at (or near) this point and causes this turbulence effect. The downwelling occurred at depths below 20 meters.

Figure 30: Vertical current velocities (mm/s) through a 700-meter cross-section of the Young Passage farm site. Data extracted from flood-tide ADCP record for the farm site, showing upwelling (blue) and down-welling (red) currents along a randomly selected centerline transect. Farm system shown by black shading; area beneath is extrapolated from data and is not considered accurate. Water depth (0-60 meters) on y-axis, and distance (meters) downstream and upstream of the farm system on the x-axis.



NOTE: *this analysis is limited to a single 'snap-shot' of the current flows across this farm site, and although providing some insight to the dynamics of these oceanographic processes, can not be considered accurate in representing the inherent complexity of such processes.*

3.4.3 Physiographic Characteristics

The physical oceanography of the Venture Point and Young Passage farm sites is determined, in a broad sense, by a shared regional tidal influence. Given that these sites are located within a few kilometers of each other, tidal exchange (elevation) and periodicity (nature of M2 constituent) are the same. However, local physiographic characteristics will define how this common tidal regime is shaped to produce site-specific circulation patterns, vertical stratification of flows, shears, turbulence, upwelling and/or down-welling, etc.. These unique physical attributes of each site will also determine how waterborne contaminant dispersion, dilution, and environmentally partitioning occur, and how these processes might affect shellfish culture performance (and tissue quality) in an integrated aquaculture system.

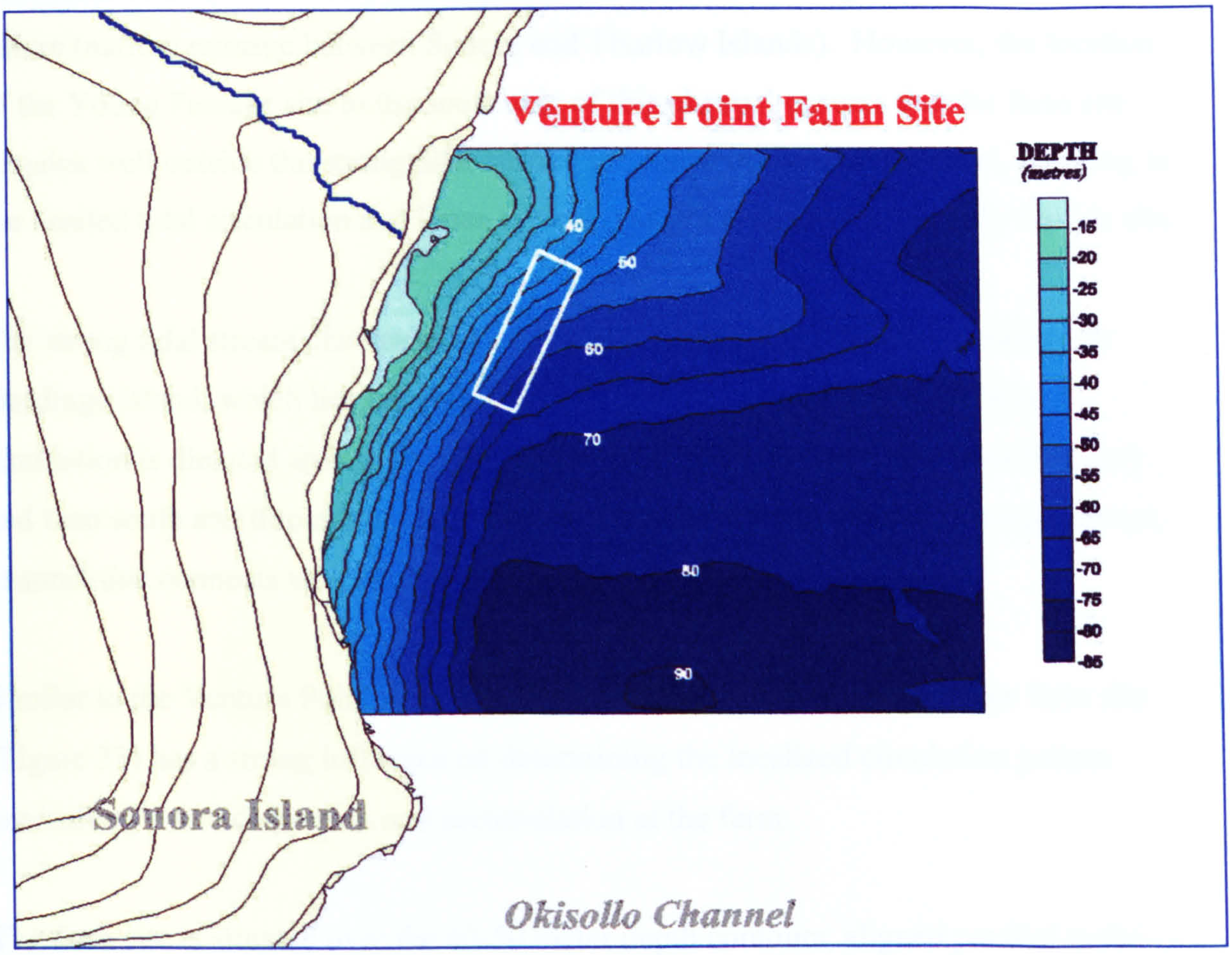
Venture Point Physiography:

Local topography of the Venture Point farm site has a direct influence on the movement of water through the study site, resulting in the dominant unidirectional circulation effect described previously. Bathymetric and substrate characteristics of a site are also attributes that will determine how a site will respond to the introduction of organic wastes, and its potential for accumulation and hence environmental persistence of such material and their associated contaminant loads.

The Venture Point farm site is located over a shallow shelf (40-60 metres) on the north side of Okisollo Channel (Figure 31). The relatively gentle-sloping shelf extends to the north and northeast of the farm, but bathymetric slope increases quickly to the southeast where it meets the centerline of the channel and depths of over 90 meters with a kilometer of the farm site. Depths beneath the net-cage system, and along the estimated dispersion pathway (downstream), range from 45-60 meters, consistent with the shelf depths on the tidal flow approaches to the farm on the north.

Figure 31:

Near-field bathymetry at Venture Point study site. Soundings derived from QTC acoustic survey conducted in Octopber/2003. The approximate position of the netcage system is indicated with the white rectangular box.



Bottom substrates at the Venture Point site are complex. Results of the QTC acoustic classification are presented in Figure 32, which suggested that six distinct substrate categories were detected from the acoustic back-scatter signals. The gradient shown in Figure 32 approximates the substrate composition of these six categories, although these results are intended as a qualitative description of the benthic environment.

The survey indicated that a soft benthic environment (shown in darker shades of brown) comprises the area described as the shelf, with mixes of silt-clay and fine sands-gravels. In contrast, the inshore (steeper) and offshore areas (to the southeast) comprise a harder substrate, composed of various combinations of gravels, cobble and boulder.

Young Passage Physiography:

The Young Passage farm site is situated in a sheltered embayment along the south side of Nodales Channel. Similar to Okisollo Channel, Nodales receives tidal flows from Johnstone Strait (to the west) and amplifies the tidal velocities given its constricted nature (narrow passage between Sonora and Thurlow Islands). However, the location of the Young Passage site to the south side of this channel ensures that the farm site remains well outside the strong tidal current influence of the main channel, resulting in the limited tidal circulation and lunar-cycle dynamics described previously for this site.

The strong tidal stream characteristic of Nodales Channel is further constrained by Hardinge Island, which lies between the farm site and the main channel. Water circulation is directed around the north side of this island (through Burgess Passage) and then south and through the farm site that lies within the southern, Young Passage, channel that connects with Nodales.

Similar to the Venture Point farm site, the bathmetry of the Young Passage farm site (Figure 33) has a strong influence on determining the localized circulation pattern controlling waste dispersion and accumulation at the farm.

The farm site is situated over the 40-50 meter depth contours, aligned parallel to the adjacent shoreline. The shallow sublittoral area (above 30 meters) is steeply sloped while the deeper water, extending offshore of the farm system, comprises a gradual slope to a maximum depth of approximately 70 meters.

The bathmetry for Young Passage also indicates a shallow sill to the west of the site (less than 25 meters), which effectively separates the flow of deepwater between the farm site and Nodales Channel through Young Passage. This feature supports the tidal circulation and lunar cycle observations that indicate a dominant (albeit slight) westerly flow through the site, suggesting that water flows around Hardinge Island, through Burgess Passage and south to the Young Passage farm site.

Figure 32: Substrate composition of Venture Point study site. Gradient based on QTC acoustic survey (October/2003) with incidental grab samples to validate composition. Bathymetric contours shown in blue, with approximate location of net-cage system in white and relative position of research longline in dotted yellow. Graph axes show latitude and longitude of study area.

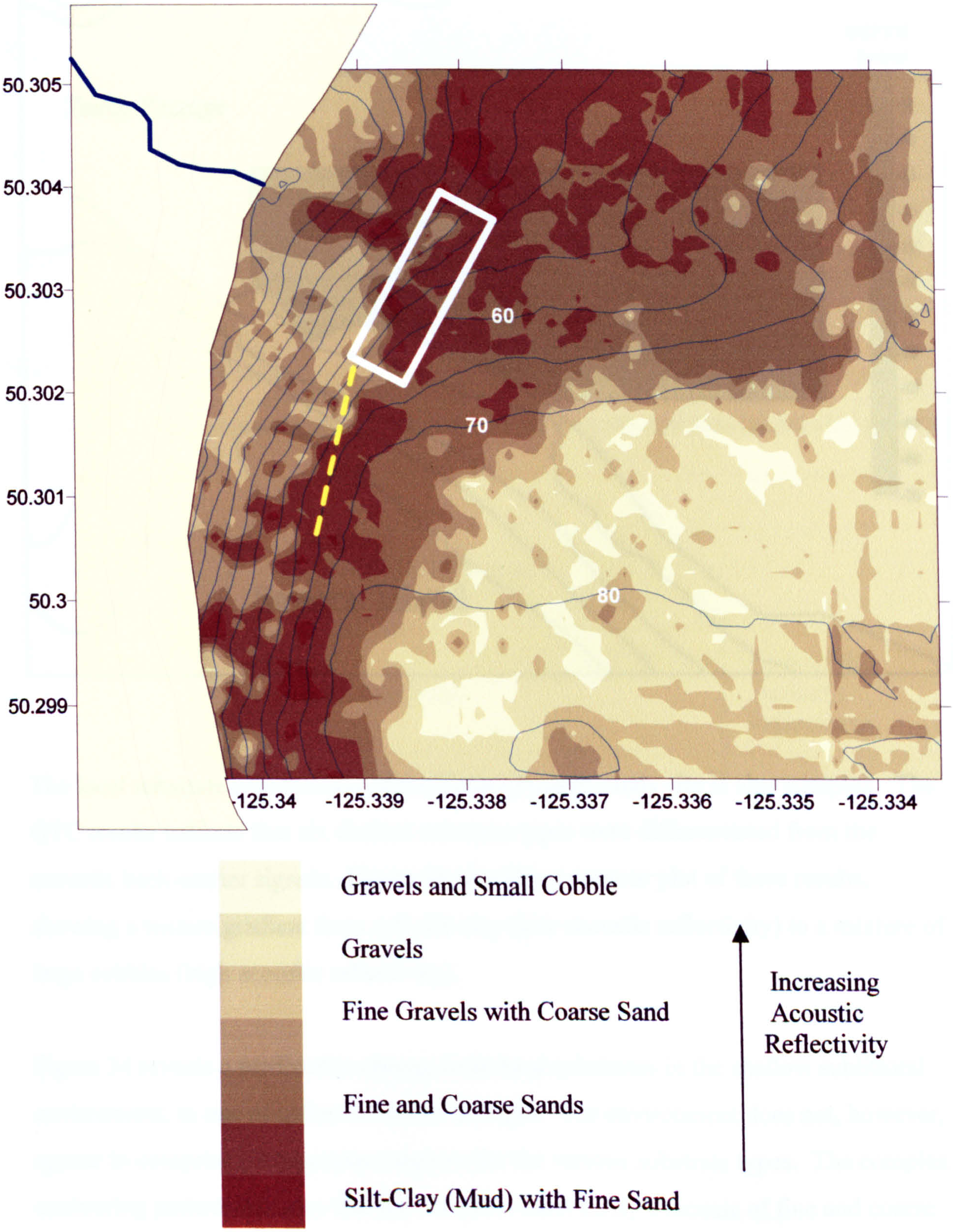
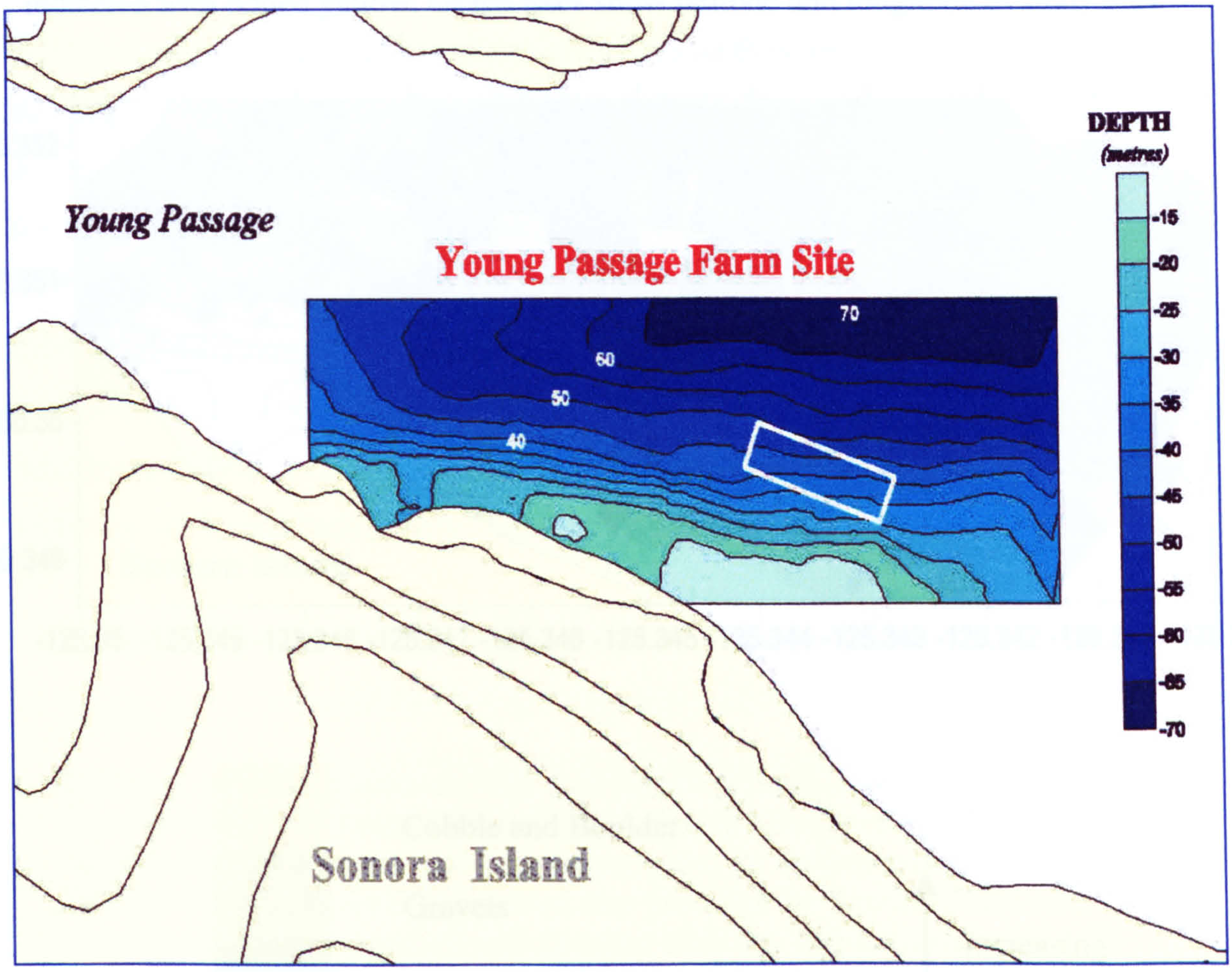


Figure 33:

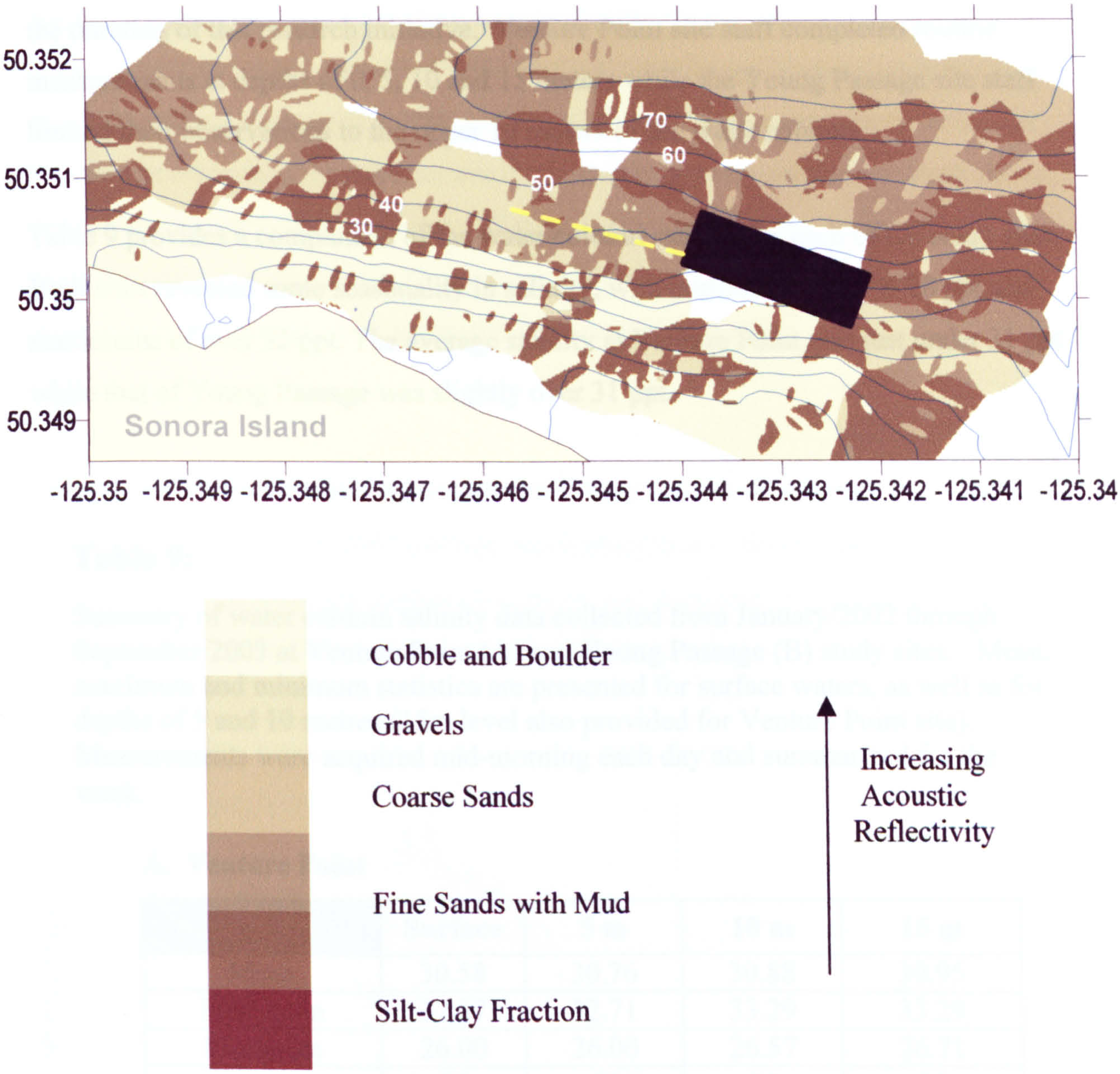
Near-field bathymetry at Young Passage study site. Soundings derived from QTC survey conducted during August, 2002. The approximate position of the netcage system is indicated with the white rectangular box.



The local substrate composition of the Young Passage study site is also complex. The QTC results indicate that six distinct substrate types were differentiated from the acoustic back-scatter signals. Figure 34 provides a contour plot of these results, showing a texture gradient from soft silt-clay (low acoustic reflectivity) to a mixture of large cobbles (high acoustic reflectivity).

Figure 34 reveals a predictable change from hard substrates in the shallow sublittoral environment, to one of softer substrates at depth. The environment does not, however, appear to comprise homogeneous regions for the various substrate types. The complex contouring pattern indicates that the bottom is most likely a mosaic of fine and coarse substrates.

Figure 34: Substrate composition of Young Passage study site. Gradient based on QTC acoustic survey (October/2003) with incidental grab samples to validate composition. Bathymetric contours shown in blue, with approximate location of net-cage system shown as a black polygon and relative position of research longline indicated with dotted yellow. Graph axes show latitude and longitude of study area.



The Young Passage farm system is situated over an area that is largely fine sediments (dark brown substrate category, Figure 34), including the downstream corridor where the experimental longline system was deployed. The low tidal energy observed at this site, in conjunction with the moderate depths, supports the apparent depositional nature of the site. Although materials are anticipated to settle and accumulate in such environments, it is equally unlikely that re-suspension of this material could occur and hence impact any water column resources.

3.4.4 Temperature and Salinity Profiles

Weekly temperature and salinity data were collected from each of the study sites over the duration of this research initiative. Venture Point site staff completed routine measurements at depths of 0, 5, 10 and 15 meters while the Young Passage site staff limited their observations to the upper 10 meters of the water column.

Table 9 provides a comparison of the water column salinity for each of the study sites. Both sites revealed some seasonality in salinity, with minimums of 26 to 28 ppt and maximums of over 32 ppt. The average salinity at Venture Point was just under 31 ppt while that of Young Passage was slightly over 31 ppt.

Table 9:

Summary of water column salinity data collected from January/2002 through September/2003 at Venture Point (A) and Young Passage (B) study sites. Mean, maximum and minimum statistics are presented for surface waters, as well as for depths of 5 and 10 metres (15m level also provided for Venture Point site). Measurements were acquired mid-morning each day and summarized for the week.

A. Venture Point

Salinity (ppt)	Surface	5 m	10 m	15 m
Mean	30.58	30.76	30.88	30.95
Maximum	32.57	32.71	33.29	33.29
Minimum	26.00	26.00	26.57	26.71
n	85	85	85	85

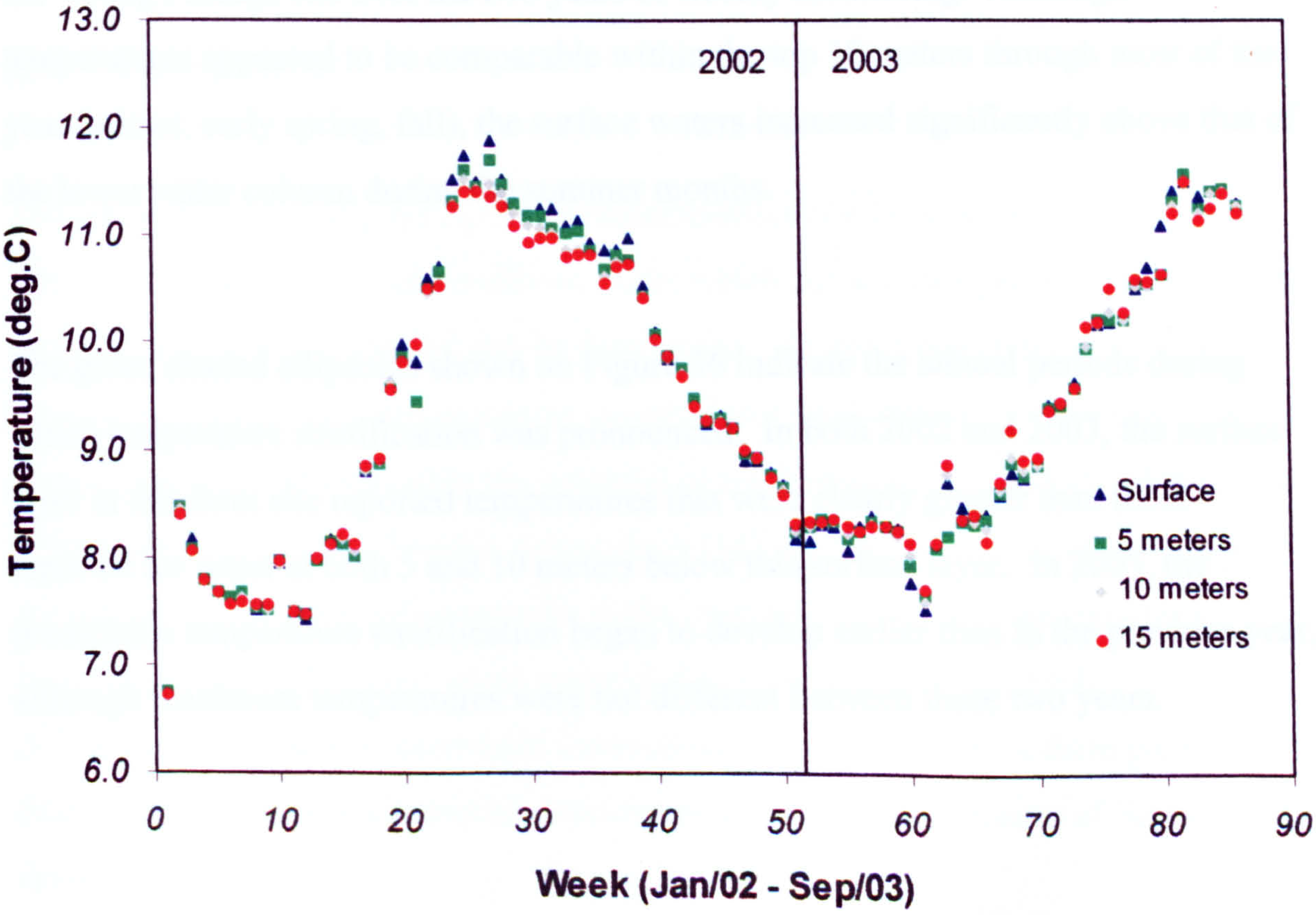
B. Young Passage

Salinity (ppt)	Surface	5 m	10 m
Mean	31.3	31.6	31.2
Maximum	32.1	32.1	32.6
Minimum	28.0	29.9	30.0
n	72	84	56

Vertical stratification in salinity was less apparent at the Venture Point farm site, with salinities consistent across all of the depths measured. Young Passage, however, revealed that the seasonal lows in salinity (28 ppt) occurred only in the surface waters (0-5 meters), with little annual change in salinity reported at depth.

Figure 35 provides a scatter plot of the weekly temperatures acquired for each of four depths (surface, 5, 10 and 15 meters) acquired at the Venture Point farm site from January/2002 through September/2003. The seasonal change in water temperatures are clearly portrayed in these data, although there is very little temperature stratification apparent in these data. In essence, water temperatures are consistent across all depths, and jointly change with the seasons. Slightly higher surface temperatures (0 and 5 meter depths) were apparent in the summer of 2002, although this minor temperature stratification was not as clear during the same period of 2003.

Figure 35: Temporal changes in water temperature at the Venture Point farm site (January/2002 through September/2003). Weekly temperatures acquired at surface (0-meters), and at depths of 5, 10, and 15 meters. Data acquired consistently in mid-morning.



In the winter, between these two high temperatures periods, a very slight inversion of temperature was noted, with the surface layer slightly cooler than that reported at depth.

The annual temperature range observed at Venture Point (Figure 35) spanned 4.5° C, with a winter low of 7.5° C and summer highs of 11.9° C. There was no apparent difference between the two years monitored.

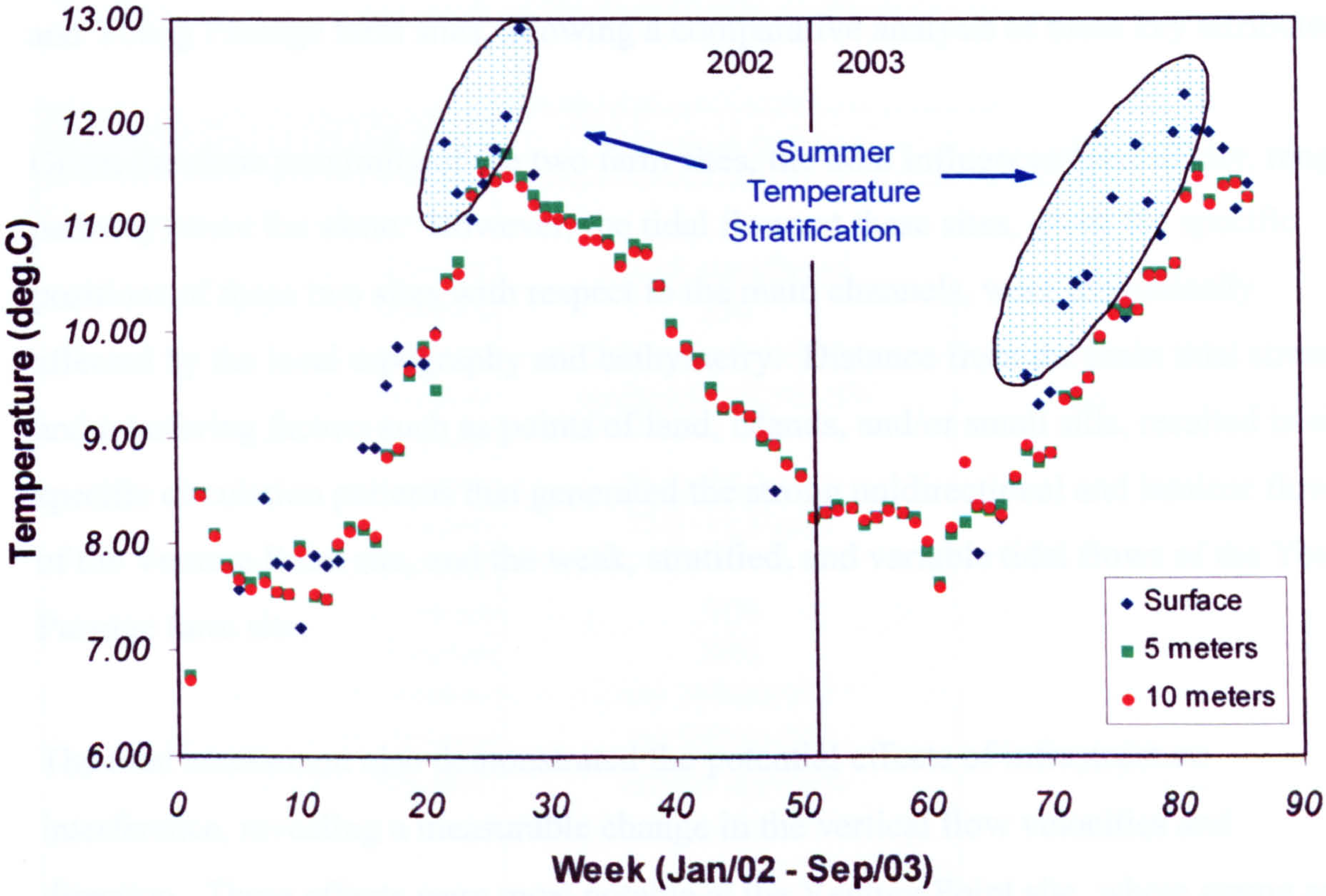
Figure 36 illustrates the weekly change in water temperatures for the Young Passage farm site over the period January/2002 through September/2003. Data acquired for the surface, 5-meter and 10-meter sample depths are plotted concurrently to document temporal fluctuations in temperature while indicating periods in which spatial differences exist.

The temperatures at the Young Passage farm site ranged from seasonal lows of 7.4° C to highs during the summer months that typically reached 12.5° C, with an annual differential of 5.1° C. As with the Venture Point site, temperature trends were the same for each of the two years monitored in this study.

Figure 36 also illustrates the extent to which temperature stratification was observed at the Young Passage site over the two years of weekly monitoring. Although temperatures appeared to be comparable within the top 10 meters through most of the year (winter, early spring, fall), the surface waters increased significantly above that of the lower water column during the summer months.

The green shaded ellipsoids shown on Figure 36 indicate the annual periods during which temperature stratification was pronounced. In both 2002 and 2003, the surface layer at this farm site reported temperatures that were clearly greater than those reported for water at both 5 and 10 meters below this surface layer. In 2003, the predictable temperature stratification began to develop earlier than in the previous year, although maximum temperatures were not different between these two years.

Figure 36: Temporal changes in water temperature at the Young Passage farm site (January/2002 through September/2003). Weekly temperatures acquired at surface (0-meters), and at depths of 5 and 10 meters. Data acquired consistently in mid-morning.



3.5 Discussion

The oceanographic and physiographic characteristics of a farm site play an important role in defining the physical conditions under which aquaculture performance (growth, survival of the culture species) can be determined. In this research program the detailed assessment of these processes is considered fundamental in achieving the primary objective of evaluating the potential of integrating finfish and shellfish species in the context of a Multi-Trophic Aquaculture (MTA) system. This study component not only provides information specific to the interpretation of the other program components (i.e., effects on: waste material dispersion/dilution, shellfish growth/survival, tissue-contaminant interactions), but also examines these physical processes in terms of site selection considerations for the development of future MTA systems.

The farm sites used in this study were selected, as discussed in Chapter 2, to represent polarized physical environments that, in a generic sense, were to comprise a site with weak oceanographic influences and one with strong such influences. Table 10 presents a summary of the oceanographic and physiographic characteristics of the Venture Point and Young Passage farm sites, allowing a comparative analysis of these key attributes.

Given the close proximity of the two farm sites, the tidal influences (periodicity, range, pathway) were the same. However, the tidal flows at these sites, given the specific positions of these two sites with respect to the main channels, were dramatically affected by the local topography and bathymetry. Distance from the main tidal stream, and interfering factors such as points of land, islands, and/or small sills, resulted in site-specific circulation patterns that generated the strong unidirectional and laminar flows of the Venture Point site, and the weak, stratified, and variable tidal flows of the Young Passage farm site.

The tidal assessment also demonstrated the potential effects of infrastructure interference, revealing a measurable change in the vertical flow velocities and direction. These effects were most notable at the Venture Point site, where strong and laminar approaching tidal flows (with a minimal vertical flow component) were physically deflected by the farm system causing significant down-welling flows to be developed on the upstream side of the farm with a displaced upwelling pattern on the downstream side of the system. At the Young Passage farm site, where tidal flows were much weaker, this interference pattern was not as clear although some upwelling was noted along the downstream side of the farm.

Vertical turbulence patterns play a major oceanographic role in the transport of momentum, mass, chemical species (nutrients, contaminants), and particles (Thomson, 1982). The small-scale turbulence observed in this study, created by infrastructure interference, may result in localized affects on water column structure, including sediment re-suspension (Chromey, 2001), plankton distribution, and particle/contaminant flocculation, dispersion and environmental persistence (Sutherland, 2001).

Table 10: Summary and comparison of oceanographic and physiographic characteristics of the Venture Point and Young Passage farm sites. Values for each parameter are summary statistics, or qualitative observation, and were derived from the data collected using each of the component methodologies described previously.

	Venture Point	Young Passage
Oceanographic Influences:		
Tidal Rhythm:	mixed, semidiurnal	mixed, semidiurnal
Tidal Range (meters):	2.92	2.92
Tidal Inputs:	via Johnstone Strait, strong	via Johnstone Strait, strong
Tidal Direction:	unidirectional; NE-SW	bi-directional; E-W
Mean Tidal Speed:		
0-15 meters	13.4 - 14.2 cm/s	7.6 - 9.9 cm/s
> 15 meters	12.3 -12.5 cm/s	7.2 - 7.6 cm/s
Quiescent Period (% time):		
0-15 meters	35%	60%
> 15 meters	50%	80%
Residal Flows - surface:	net 200 km SW	net 50 km W
Residual Flows - bottom:	net 200 km SW	net 30 kn E
Vertical Flow Structure:	laminar approach strong system effects 50 meters upstream 150 meters downstream	laminar approach weak system effects 0 meters upstream 50 meters downstream
Salinity (ppt):	26 - 32	26 - 32
Temperature (deg.C):	7.5 - 11.9	7.4 - 12.5
Stratification:	none	summer, temperature; winter, salinity
Physiographic Influences:		
Main channel alignment:	east-west	east-west
Farm position with respect to main channel flows:	side of main channel	well off channel, embayment
Topgraphical influences:	point of land	island, embayment
Depth at site:	45-50 meters	45-50 meters
Bathymetric influences:	sloping non-depositional	sill to west; limits bottom flows; gentle slope depositional
Bottom substrate:	variable; sands through rock	variable; finer sediments

The spatial and temporal variability of these processes will be determined by site-specific oceanographic and physiographic attributes (as described for the two study sites), which could also influence the engineering design of MTA infrastructure. The relationship between system design and the environmental attributes of a site (tidal flow, bathymetry) could be used to maintain optimal conditions for aquaculture performance (health and productivity), taking advantage of sustained water quality, and optimal phytoplankton and/or seston flux (food resource for shellfish component) to the system.

An MTA system that considers these oceanographic characteristics in an optimal design would also, presumably, minimize the potential negative effects of waste products (particulate and dissolved) that would be released to the water column from the finfish component of the system. Increasing flow velocity and vertical mixing would result in an increase in waste dispersion with a commensurate reduction in the localized persistence of such contaminant loads. Sites with unidirectional flows, such as Venture Point, would also eliminate the return component of the tidal cycle, resulting in high residual flows, that would result in a further amplification of the benefits associated with a well-flushed farm site.

In contrast to a farm site with strong oceanographic influences (such as Venture Point), and the inherent benefits that could be realized in an MTA system that considered these attributes in its design and engineering, a site with sub-optimal water exchange will suffer the negative consequences of lacking such influences. In the case of the Young Passage farm site, the tidal fluctuations were generally weak and bi-directional in nature. With current velocities in a quiescent state for between 50 and 80% of the time, the exchange of water is significantly reduced (much lower residual flows) and the potential for increased contaminant residency (or persistence) much higher within the water column (Sutherland, 2001) as well as in the sediments as particulates settle under these quiescent conditions (Milligan and Loring, 1997).

These site characteristics could also negatively impact the productivity and health of each component of an MTA system, with implications on the economic viability of such a system. In addition, contaminant persistence resulting from limited circulation could affect tissue quality of the culture component(s) and potentially represent a human health (consumption) risk.

As suggested by this discussion, the oceanographic and physiographic attributes of a farm site will play an important role in determining the appropriateness of a site to support not only monoculture (Cross, 1993), but for a Multi-Trophic Aquaculture (MTA) system. Site selection is the key to ensuring economic viability (optimal production, sustained system health) as well as for optimizing water quality conditions through efficient waste assimilation.

The oceanographic and physiographic data acquired for each of the study sites will assist in the subsequent interpretation of results from the other components of this research project. These processes will determine the extent of organic waste dispersion and accumulation (Chapter 4), the persistence of waterborne contaminants and bioavailability to suspended shellfish (Chapter 5), and the shellfish culture performance assessed for each of the study sites (Chapter 6).

CHAPTER 4

Contaminant Inputs and Dispersion

4.1 Introduction

There have been numerous scientific studies, over the past 15-20 years, that have examined the environmental effects of organic waste loading associated with marine finfish aquaculture facilities (Ackefors and Enell, 1990; BCEAO, 1997; Gowan and Bradbury, 1987; Hargrave, 2003; Levings, 1994; Morrisey et al., 2000; Rosenthal et al., 1995; Weston, 1996). These international research initiatives have reviewed, measured, and modeled the organic waste fluxes from finfish farm facilities, the physical-chemical processes affecting assimilation (and persistence) of these wastes, the disruptive or destructive impacts this material has to resident biological communities, and the spatial extent of these impacts around these operations.

In terms of spatial impacts, a focus on the near-field effects resulting from accumulation of organic wastes have documented the sediment chemistry and geophysical processes involved in the assimilation of organic waste inputs (Brooks, 2001; Hargrave et al., 1997; Wildish et al., 1999). Nutrient loading from farm operations have been implicated in potential eutrophication processes, with speculation that harmful algal blooms can be generated and sustained as a result of the cumulative nutrient enrichment (far-field) effects from such aquaculture facilities (BCEAO, 1997; Whyte et al., 1999).

Despite the considerable effort that has been dedicated to investigating the fate and effects of the *'total organic wastes'* generated from marine net-cage facilities, comparatively little attention has been placed on the evaluation of the potential environmental impacts of the residual chemical constituents of these wastes (Chou et al., 2002; Burridge et al., 1999). Micronutrients of fish feed, disinfectants, antibiotics, chemotherapeutic compounds, anaesthetics, and antifoulants are all released to the marine environment, albeit typically in very low concentrations and at differing release rates.

The trace metal constituents of fish feed represent a continual flux to the receiving environment, with release of the residual fraction of these dietary metals in faecal material and in any wasted (uneaten) feed pellets (Chou et al., 2002). The mineral component of fish feed comprises a number of key elements (calcium, sodium, magnesium, potassium, phosphorus, chlorine and sulphur) are considered critical to the structural development of tissues and bone, and also play important roles in osmoregulation (Jobling, 2001). Minor trace metals such as manganese, zinc, copper, selenium and cobalt are all required for a variety of metabolic processes (Desilva and Anderson, 1995).

Although a few of these elements are added to the feed during the milling process to satisfy these specific physiological requirements of the fish (Lall, 1991; Tacon and Desilva, 1983), most are associated with feed components (e.g., fish meal) and will vary in composition and concentrations over time, and with source (Lorentzen and Maage, 1999). Zinc, for example, is incorporated into the feed and serves not only as an essential metalloenzyme (Desilva and Anderson, 1995) but it also reduces the risk of inducing cataracts in juvenile salmon (Richardson et al., 1986). Other elements, such as cadmium or mercury, are subject to bioconcentration in wild fish stocks (source of fish meal for meal) and can vary significantly among regions, and even locally (Hamilton, 1989).

The use of antifoulant compounds represents an additional pathway for trace metal release to the marine environment (Chou et al., 2002; Cross, 2003). The active constituent in the majority of these commercial products is copper oxide, which is applied to the surface of nets and provides a slow and continuous release of ionic copper that is toxic to the fouling organisms that may attempt to settle on these structures. In Scotland, for example, there are currently 25 commercial antifoulants that are licensed, or provisionally licensed, by the Scottish Health and Safety Executive for use in Scottish aquaculture (SEPA, 2000). All salmon growing jurisdictions use antifoulants of some kind, and most remain copper oxide based.

The use of therapeutic compounds in the treatment and/or the prevention of bacterial infections is prevalent throughout the marine finfish aquaculture industry, although significant improvements in husbandry practices over the past 10 years have resulted in a dramatic decline in required usage of these compounds. Nonetheless, periodic outbreaks of bacterial diseases are a risk of confined animal rearing. The typical application of such compounds involves the introduction of medicated feed in place of regular feed, with the antibiotic absorbed by the fish during the digestive process.

Oxytetracycline (OTC) is a generic antibiotic that is employed universally in poultry, swine and cattle farming as well as in finfish aquaculture to prevent or treat a variety of bacterial infections. In British Columbia, oxytetracycline made up 89.6% of medications given in feed in 1995 (Hewitt, 1997). Given its continued role as the dominant antibiotic used by the industry in all regions, research using it as the representative for this class of feed constituents is appropriate.

HPLC detection methods have long been established for detecting OTC residues in finfish flesh (Reimer and Young, 1990; Iwakai et al., 1992; Agasoster, 1992a, 1992b; Long, 1990). Kinetics of tissue uptake and elimination in salmonids have been studied by Namdari et al. (1996), Bjorklund and Bylund (1990), Martinsen (1995), Nordlander, (1987), Elema et al. (1996), and by Aoyama et al. (1991).

The potential bioavailability and seafood safety consequences of these antibiotic compounds to non-target species in the surrounding marine environment has been identified as one of the major environmental concerns associated with marine netcage culture on this coast (BCEAO, 1997). The possible transfer of these antibiotic compounds, and their residues, to wild fish and/or to species of other trophic levels is the basis of concern, with the dispersion and persistence of these compounds considered of importance in the assessment of environmental impacts of salmon aquaculture.

OTC is typically delivered to fish orally, by incorporating this therapeutic compound into the feed. In Canada oxytetracycline may be administered in fish feed at a rate of 75 mg/kg/day for 7 to 10 days, with a withdrawal period of 30-60 days. Under veterinary prescription the dose may be higher (e.g., 100 mg/kg/day) but it must be accompanied by an extended withdrawal period. Maximum residue levels (MRL's) for fish flesh are set by the Feeds Act (Canada) and are currently at an assigned level of 0.1 ppm. In the US, this criterion is twenty fold higher, at 2.0 ppm.

Oral OTC has a low apparent oral bioavailability to the fish (Bjorklund and Bylund, 1991) and, depending on diet composition, may pass through the fish gastrointestinal tract mostly unmetabolized (Cravedi, 1987; Luzzano et al, 1994). Previous studies using oral applications have suggested that as much as 70-80% of the applied drug (deployed in feed) may end up in the sediments beneath the farm structures (Samuelsen et al, 1992). This may be due to either unused feed or excretion of fecal material. OTC binds to particulate matter that is deposited on sediments underneath the farm (Lunestad, 1992).

Several studies have questioned the biological significance of detecting OTC in sediments or seawater. Hektoen et al, (1995) showed that depuration of OTC out of sediments is most likely due to leaching and redistribution from surface sediments rather than degradation. Once in solution OTC binds heavily (95%) to divalent cations, in particular magnesium (Hektoen et al, 1995; Lunestad and Goksyor, 1990). Experimental studies have shown that such binding may significantly decrease its antibacterial activity (Barnes et al, 1995; Lunestad, 1990), particularly in the presence of salt concentrations approximating seawater. OTC is also degraded by light at depth and shows reduced antimicrobial activity (Lunestad et al, 1995). Smith and Samuelsen, (1996), in their 1996 model kinetics for OTC re-entry into the water column concluded that resultant OTC concentrations in the presence of magnesium and calcium were unlikely to be of biological significance.

Chemotherapeutic compounds represent yet a third group of potential contaminant constituents originating from feed used at finfish aquaculture farms. As with antibiotics, many of the chemotherapeutic compounds are milled directly into the feed, at a prescribed dosage, and delivered to the fish as a medicated treatment when required of a production cycle. A common example of this class of compounds is the pesticide emamectin benzoate, which is offered as the commercial sea lice treatment drug Slice™.

Emamectin benzoate, or Slice ™, has been marketed as an effective treatment against juvenile, motile pre-adult, and adult life stages of the sea lice species *Lepeophtheirus salmonis* (Schering-Plough, 2002) and *Caligus elongatus* (SAMS, 2003). These ecotoparasitic copepods attach to salmonid, and other marine fish species, leaving open lesions that become susceptible to bacterial or viral infections that can cause mortality in a culture population.

Salmon readily absorb emamectin benzoate from the medicated feed, and release to the environment typically occurs through faecal material or via wasted (uneaten) feed. Excretion of the metabolites occurs for an extended, post-treatment period, and given the insoluble nature of the compound and its affinity to particulate matter, most of this waste material settles to the seafloor where it becomes a localized risk to the epifaunal and infaunal benthos (SAMS, 2003).

SEPA (1999) conducted a thorough environmental risk assessment for emamectin benzoate and also concluded that the compound is likely to remain tightly bound in sediments due to its low seawater solubility. The study further summarized the break-down and degradation characteristics of the compound in sediments, suggesting that under aerobic conditions the half-life was 193.5 days and approximately 427 days under anaerobic conditions. However, in the presence of light and contact with microbially active sediments the half-life was as little as 5 days.

The leaching of copper from antifoulant-treated netting, and the release of the residual trace metals fraction associated with feed inputs, represent low-level waterborne contaminants that could have a significant effect on any potential MTA component that is considered for development within the zone of influence of these waste constituents. Similarly, the periodic use of chemotherapeutic compounds, including both antibiotics and sea lice treatment pesticides, may have negative effects on species that are co-cultured with a finfish component. Given these concerns, an understanding of the waste constituent fluxes, and their spatial/temporal dispersion patterns is essential in considering the technical feasibility of developing MTA.

4.2 Objectives

The objective of this study component was to document the organic waste contaminant composition and dispersion pattern for each of the study sites. The specific objectives for this evaluation included:

- Measurement of the organic waste flux along the dispersion pathway extending downstream of each study site;
- Classification of the chemical constituents of the organic waste, including an estimate of the temporal/spatial variability in these constituents; and
- Analysis of the organic waste loading in relation to feed inputs, faecal material generation/discharge, and estimation of the proportion of organic wastes (and their chemical constituents) that remain unaccounted for in this mass balance assessment.

The results of this study component are subsequently used in a discussion of potential contaminant inputs, spatial and temporal distribution patterns related to differing oceanographic attributes of the study sites, and the estimated environmental partitioning (dissolved versus particulate constituents) and the potential impacts on non-target organisms (shellfish) located in close proximity of the cage systems. A discussion of the implications of these processes on the development of Multi-Trophic Aquaculture (MTA) is also presented.

4.3 Materials and Methods

The assessment of organic waste material attributes (contaminant constituents), dispersion, accumulation, and its environmental bioavailability (spatial and temporal dynamics) was completed using a combination of sample collection, chemical analyses, and data evaluation procedures. The following sections describe these approaches for this study component.

4.3.1 Farm Operational Data

This research project was conducted using facilities operated by Marine Harvest Canada (Young Passage farm site) and Heritage Salmon Limited (Venture Point farm site). Each company maintains a computer database management system for its fish husbandry information, which includes all operational aspects over each production cycle. Daily information is acquired from the farm site and logged onto the on-site database. Weekly compilations are sent to the main corporate offices where they are consolidated into a common operational database.

For this research initiative each participating company provided access to their main database and permitted records associated with the production cycle for the respective study sites to be downloaded and used for analytical purposes. The records acquired for this study component included:

- Weekly fish production statistics (number of pieces, biomass, FCR's);
- Weekly fish Mortality;
- Feed deployment records (quantity);
- Harvest logs;
- Net change logs;
- Chemotherapeutic application (and withdrawal) records; and
- Environmental (temperature, salinity, dissolved oxygen) reports.

These data were used in a comparative analysis with field data acquired on waste dispersion and accumulation in an effort to relate operational dynamics with environmental response.

4.3.2 Waste Material Loading

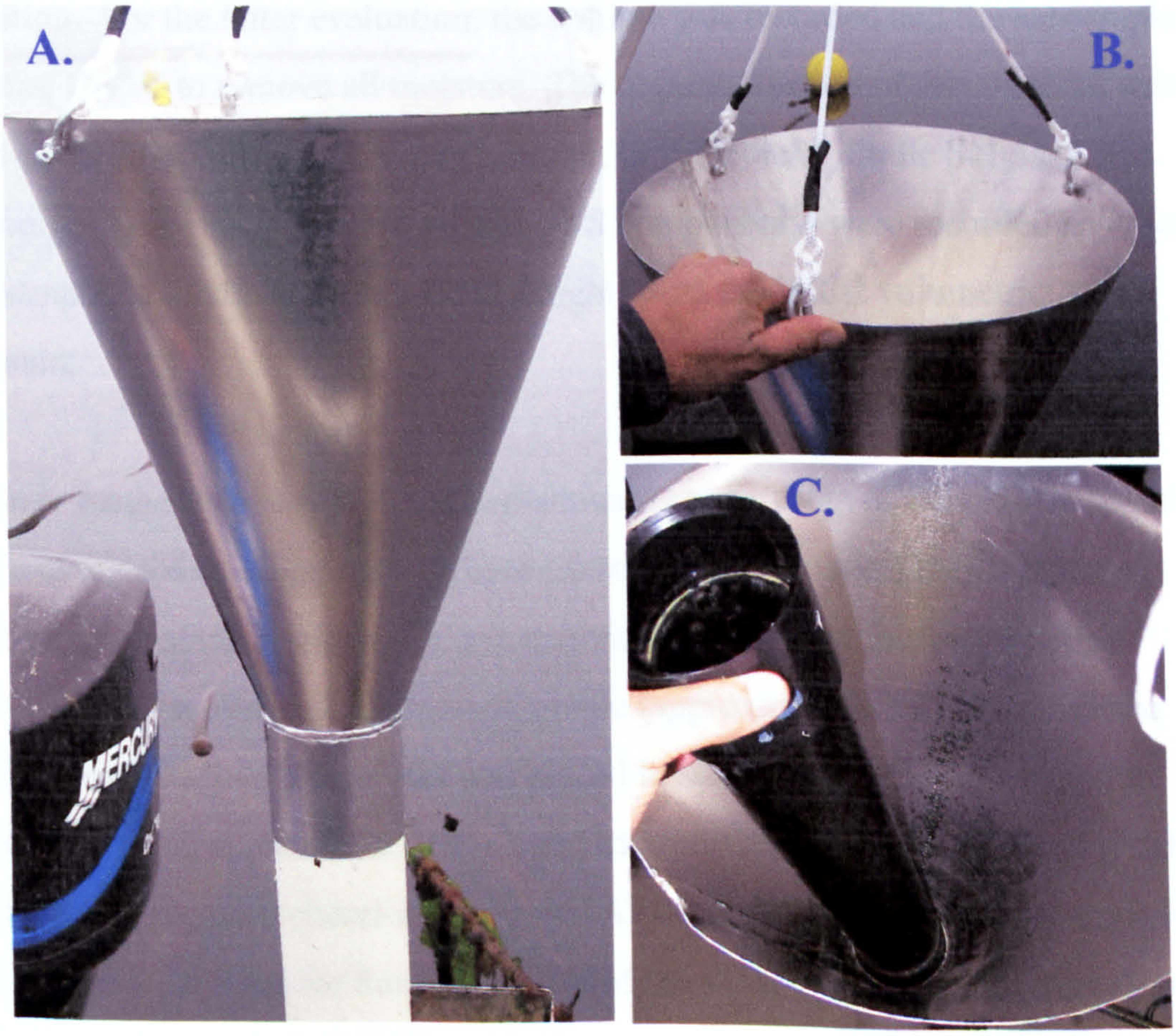
Sediment collection canisters, or waste traps, were deployed in pairs on a single dropline for each of the ten downstream stations established for the two study farm sites. As stated previously (Study Design – Chapter 2), the canisters were placed at two depths: (i) 15 meters below the sea surface; and (ii) at 5 meters above the sea floor at Higher High Water (HHW). The 5-meter clearance of the bottom canister allowed for vertical movement during the tidal cycle (2.92 meter exchange), yet was considered representative of the seafloor in terms of current flow (quiescence). The positions of each canister, given that they were secured from the floating longline system, remained in the same relative position as the floating finfish farm system and thus the point of waste release from the composite net-cages.

The sediment canisters were manufactured using a combination of PVC piping and aluminum sheeting (Figure 37). The overall height of each unit was 110 cm. The lower portion of the unit consisted of a PVC cylinder, approximately 60 cm in length, and was connected (and sealed) to the upper aluminum capture cone. The aluminum capture cone was designed to have a large capture area (45 cm diameter; 0.16 m^2).

To facilitate retrieval of material collected by these units, an internal PVC sleeve (Collection Chamber) was developed that fit tightly into the bottom portion of each unit (Figure 37-C). This allowed retrieval and replacement of the Collection Chamber inserts in the field, allowing for processing of the material under conditions that prevented material loss through spillage. Each Collection Chamber was equipped with a plastic cap, and was labeled with site and sampling station information to ensure proper processing of acquired samples.

Figure 37: Sediment collection canister used in study.

A: side view of sediment collection canister.
B: capture cone of canister (45 cm diameter) showing suspension harness.
C: collection chamber insert being place into system; top edge beveled to prevent material build-up.



Sediment canisters were deployed for a length of time considered sufficient to acquire an adequate volume of material for the subsequent analytical procedures (approximately 30 days). It was anticipated that the far-field sampling stations would accumulate minimal (if any) organic waste material, and the duration of the deployments was thus determined by the remote stations rather than those located in the vicinity of the net-cage system.

During each sampling period the canister inserts were removed, capped, and transported (in their entirety) back to the laboratory for processing and sub-sample analysis. A new insert replaced the one removed (Figure 37-C), allowing the sampling program to continue through the production cycle without interruption.

4.3.3 Organic Waste Composition

Retrieved sediment canisters (inserts) were processed in the laboratory and subsamples extracted for a number of physical and chemical analyses. Each sample was allowed to settle for 15 minutes and the volume of solids measured. A subsample of this material was retained for chemical analyses and a second sample set aside for biomass determination. For the latter evaluation, the volume was recorded and the subsample then dried at 104° C to remove all moisture. The organic fraction of this material was estimated as the sample weight Lost on Ignition, or the Total Volatile Solids (TVS). The dry weight of the subsample (total and TVS components) were recorded and then used to extrapolate total collected sample weight using the initial volumetric measurement.

As this study focused primarily on the contaminant constituents of the organic waste material, and the potential impacts of these constituents on non-target ecosystem components, most of the retained canister material was used for this characterize purpose. Thus, upon retrieval, a large subsample (approximately 250 ml) extracted directly from each sediment cansister was placed in sterilized glassware, packed in coolers maintained at approximately 4° C, and transported to a third-party analytical laboratory for the required chemical analyses. ALS Laboratories of Vancouver, British Columbia, were contracted for this component of the study given their accreditation with Canadian regulatory agencies (including Canadian Food Inspection Agency - CFIA). It was anticipated that the involvement of such a facility ensured that results from these chemical analyses would be accepted by regulatory agencies. In particular, these results could have some influence on decisions regarding required changes to regulation that would permit integrated finfish-shellfish aquaculture in the future.

The chemical constituent characterization of the organic waste samples included routine analysis (45-day sampling interval) for a suite of trace metals (25 metals; see Table 1, Chapter 2), incidental post-treatment evaluation of antibiotic residues (oxytetracycline and tetracycline) and pesticide treatment for sea lice (emamectin benzoate). The analytical procedures employed for these analyses are summarized as follows.

Trace Metals

These analyses were carried out using procedures adapted from "Recommended Guidelines for Measuring Metals in Puget Sound Marine Water, Sediment, and Tissue Samples" prepared for the United States Environmental Protection Agency and the Puget Sound Water Quality Authority, 1995. Sediment samples were homogenized either mechanically or manually prior to a heated digestion with nitric acid and hydrogen peroxide. The digest was subjected to analysis by inductively coupled plasma - mass spectrometry, inductively coupled plasma - optical emission spectrometry, or atomic absorption spectroscopy.

Oxytetracycline (OTC) & Tetracycline Residues

The specific details of this procedure were developed, and remain proprietary, to ALS Analytical Laboratories. In general, these analyses were carried out using procedures adapted from "Official Methods of Analysis of AOAC International", 16th Edition, AOAC Official Method 995.09, "Chlortetracycline, Oxytetracycline, and Tetracycline in Edible Animal Tissues" for the use with an organically-rich sediment sample. Sediment samples were extracted with a pH 4 buffer and the crude extract then cleaned up on C_{18} solid-phase extraction column prior to analysis by liquid chromatography using a C_{18} column and ultraviolet detection.

Emamectin benzoate

Emamectin benozoate analyses were conducted using procedures adapted from Yoshiti (2000) and from Riet et al. (2000) that describe an analytical approach for quantifying the chemical constituents of the avermectin family of pesticides. In general, these analyses comprised an extraction stage (using acetonitrile), clean-up on a C_{18} solid-phase extraction column, and analysis using a high performance liquid chromatography (HPLC) with fluorescence detection. Specifics of this analytical approach, including detection wavelength, were requested to remain proprietary by the analytical laboratory.

4.3.4 Data Analysis

Data acquired from this component of the research program were compiled in a working spreadsheet, for each study site, and subsequently extracted and summarized in a variety of tables and figures that illustrated waste flux with respect to distance from the respective net-cage facilities. Spatial differences in waste flux were also compared between mid-water samples and those collected from just above the seafloor. In all cases, data from the various deployment periods were standardized as daily waste inputs and the eight sets of waste canister results used as sample replicates to provide a smoothing function to the otherwise variable data.

The chemical constituents analyses conducted on the waste material were also examined in terms of flux, with average results used to assess the spatial distribution (vertically and horizontally) of these contaminant loads at the two study sites. An effort to account for the contribution of these constituents to the particulate and dissolved distribution pathways was attempted using feed compositional data, documented levels in fish tissue, and the results of this study. A crude estimate of the partitioning of these constituents was presented and discussed in terms of potential effects on species of an MTA system.

4.4 Results

The waste flux and compositional data acquired from the sediment canister component of this study were used to assess the dispersion, accumulation, and to determine the potential bioavailability of these waste constituents to shellfish grown in close proximity to these contaminant sources.

4.4.1 Farm Production

The Young Passage farm site received approximately 1.0 million smolts, averaging 180 grams, during the spring of 2002. Initial growth of these fish through December 2002 preceded grading and fish transfers to other farm sites, ultimately leaving approximately 500,000 1.0 kg fish for on-growing. Harvest was initiated in September/2003, with an average fish weight of 2.85 kg (Chinook salmon, *Oncorhynchus tshawytscha)*.

At the Venture Point farm site smolt entry occurred following a 1-year fallow period. Atlantic salmon (*Salmo salar*) smolt entry (60-gram) was staggered over a number of months, continuing until a total of approximately 1.0 million fish had been received at the site. At the end of 2002 the site was graded and half of the fish moved to another site. The remaining 0.5 million fish were grown out to harvest size (approx. 3.0-3.5 kg) over the subsequent 7-8 months.

Fish mortality over each production cycle was reportedly within acceptable operational limits (<10%), with no unusual (prolonged) periods of stress that could affect fish culture performance (e.g., phytoplankton bloom events).

During the study period for this research component, conducted from late 2002 through until the end of the field program in September/2003, the production cycles at the farm sites were closely synchronized in terms of fish entry timing, grading, on-growing and harvest timing. Despite the inherent differences between species of fish, governing growth rates and feed consumption (including conversion rates), the operation of each farm provided a direct comparison of waste production, dispersion and assimilation without the need to adjust for the timing of the production cycle.

4.4.2 Feeding Chronology, Waste Flux and Waste Dispersion

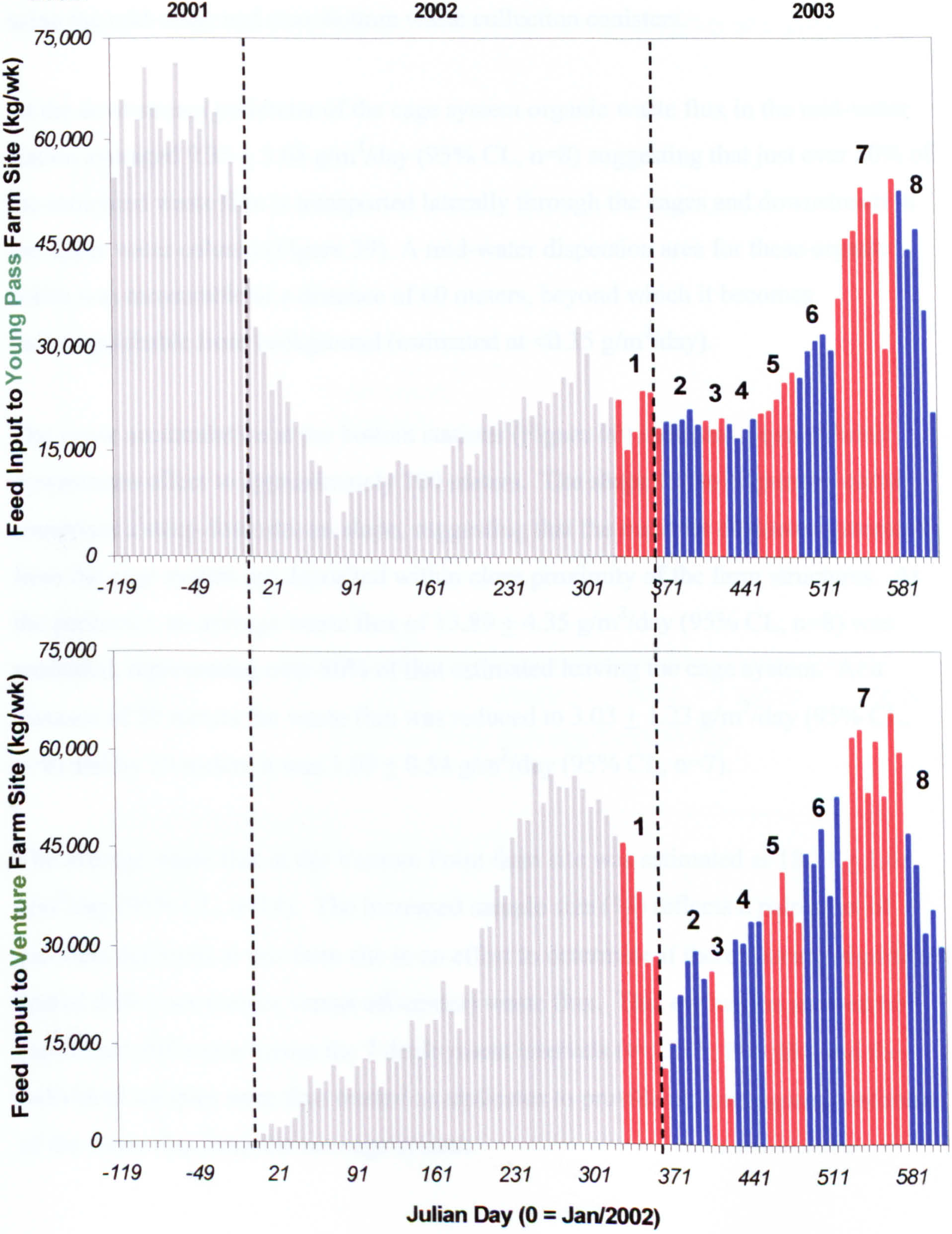
The feed records for the Young Passage and Venture Point farm sites are presented in Figure 38. These data represent weekly totals (kg feed dry weight) from late 2001 through September/2003, and thus through until the end of this study.

The feeding history for these study sites is illustrated with the gray bars (Figure 38), which also indicate the production status for these sites prior to the deployment of the research infrastructure (including sediment collection canisters). Venture Point was in fallow prior to January/2002 while the Young Passage site supported continual production through this entire period. By January/2003, the production cycle of the two sites were close to synchronized, with weekly feed input of approximately 15,000 kg/week. This showed a continual increase over the respective growth cycle, culminating in feed inputs of approximately 55,000 kg/week at Young Passage and just over 60,000 kg/week at the Venture Point farm site. The differences in feed input reflect growth rates of the two species of salmon, and the higher end-weight of the Venture Point fish (*Salmo salar*) at the time of harvest (August-September, 2003).

Figure 38 also indicates the period over which sediment collection canisters were deployed at each farm site. The grouped coloured bars (red and then blue) show the exact deployment period for each of the eight waste collection intervals. The deployment, recovery and processing of samples was coordinated between the two study sites, ensuring consistency in the environmental covariables that could affect waste dispersion, settling rates, flocculation, dissolution, etc. During this 10-month portion of the study a total of 1.15 tonnes of feed was used at the Young Passage farm site (95,862 kg/cage). At the Venture Point site the fish were fed 1.51 tonnes, or approximately 125,750 kg/cage.

Figure 38:

Total weekly feed input (kg dry wt.) for Young Passage and Venture Point farm sites over study period. Weeks over which synchronized waste collection canisters were deployed are shown with alternating series of red and blue bars – 8 assessment periods. Venture Point was in fallow prior to fish entry in January 2002.



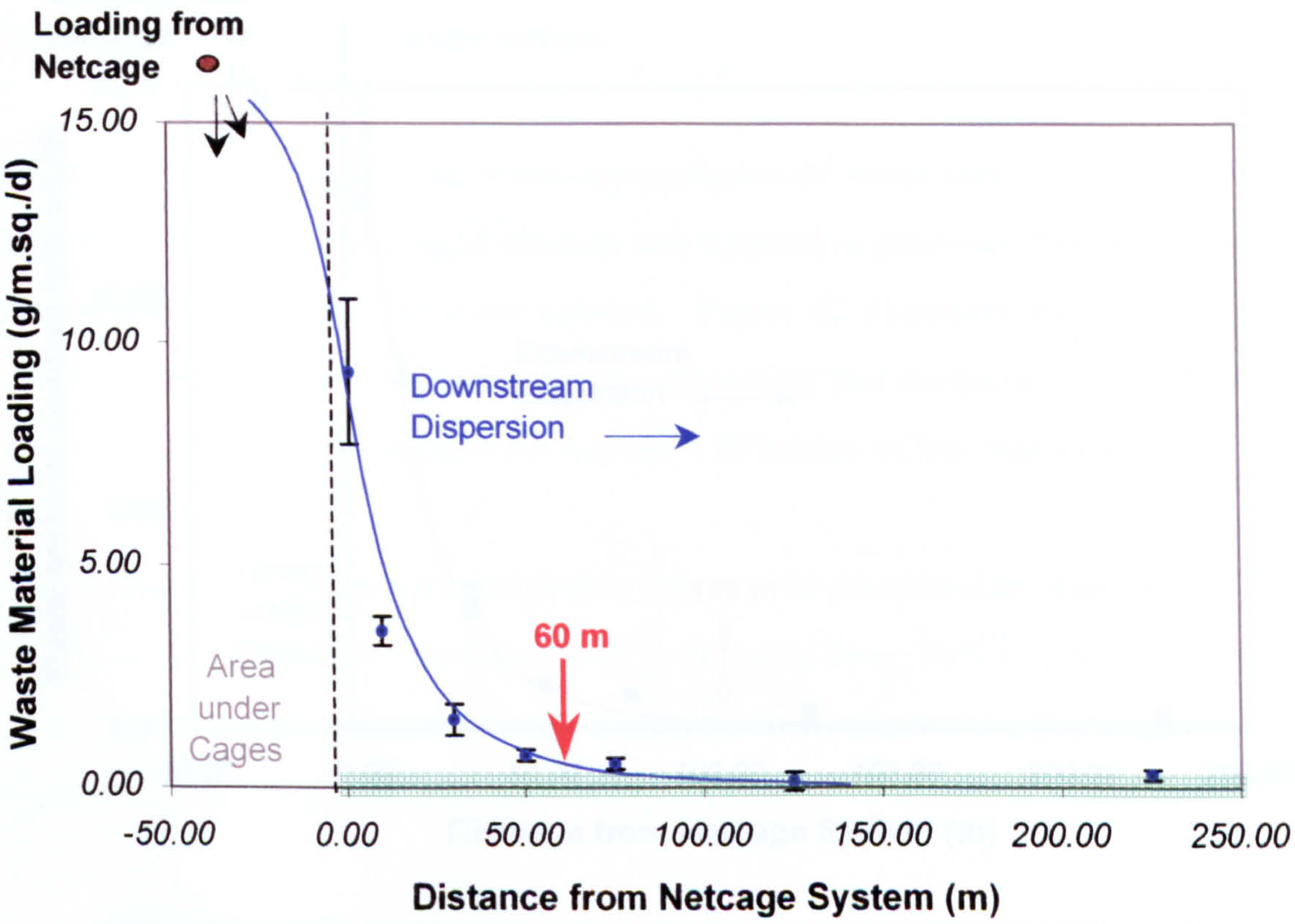
The organic waste material inputs from the net-cage system at the Young Passage farm site, measured in the mid-water column and at the seafloor, are illustrated in Figures 39 and 40, respectively. An estimate of the loss directly at the test cage was provided using the canisters deployed at the bottom (inside) of the cage system. An average organic waste flux of 17.11± 2.80 $g/m^2/day$ (95% CL, n=9) was estimated at this in-cage station, providing the basis for comparison of the downstream distribution pattern using the mid-water and near-bottom waste collection canisters.

At the downstream perimeter of the cage system organic waste flux in the mid-water station averaged 9.35 ± 5.08 $g/m^2/day$ (95% CL, n=8) suggesting that just over 50% of the estimated waste flux is transported laterally through the cages and downstream in the upper water column (Figure 39). A mid-water dispersion area for these organic solids was measurable to a distance of 60 meters, beyond which it becomes indistinguishable from background (estimated at <0.35 $g/m^2/day$).

The waste accumulation at the bottom stations (Figure 40) indicates a quantifiable downstream effect to approximately 100 meters. The shape of the dispersion curve comprises a steep downstream slope, suggesting that the majority of wastes generated from the cage system are deposited within close proximity of the farm structures. At the perimeter, an average waste flux of 13.89 ± 4.35 $g/m^2/day$ (95% CL, n=8) was measured, representing over 80% of that estimated leaving the cage system. At a distance of 30 meters the waste flux was reduced to 3.03 ± 1.23 $g/m^2/day$ (95% CL, n=8) and by 50 meters it was 1.09 ± 0.54 $g/m^2/day$ (95% CL, n=7).

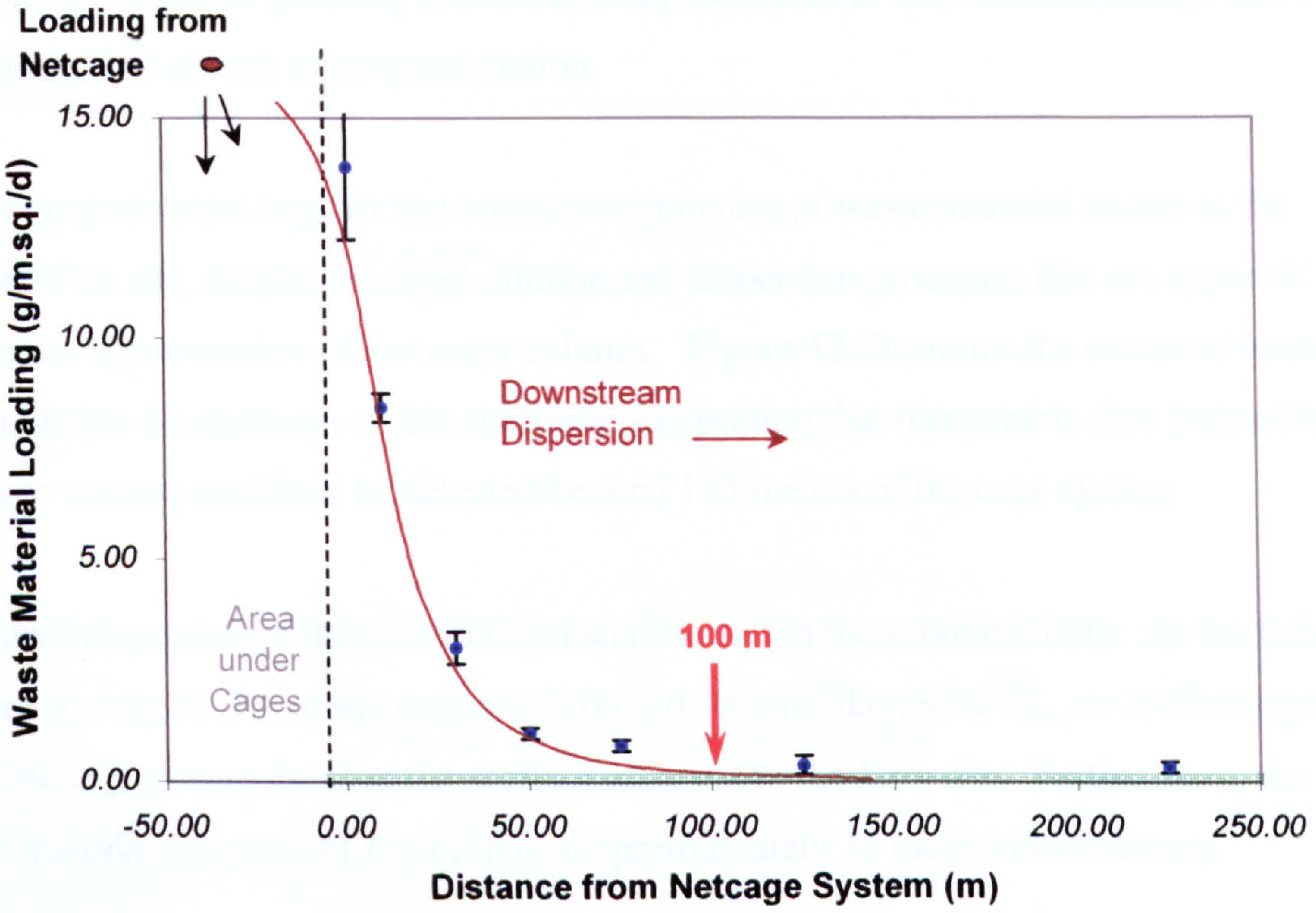
The average waste flux at the Venture Point farm site was estimated at 18.31 ± 1.61 $g/m^2/day$ (95% CL, n=14). The increased sample size (14) reflects a paired set of canisters deployed at this farm site in an effort to determine if there was any significant spatial difference (center versus off-center) waste flux. This evaluation showed no significant difference across the 7 deployment intervals (t=1.215; P<0.05), and the individual samples were thus treated as replicates to provide a more rigorous estimate of the waste flux from the net cage system.

Figure 39: Organic waste material loading (g/m^2/day) at Young Passage farm site – Midwater Canister Data. Average inputs (lower table) shown graphically in spatial relationship with netcage system and with reference to background levels of sedimentation (green band is mean ± 95% CI). Red arrow indicates distance beyond which farm wastes can not be differentiated from background flux of organic material.



		Total Waste *(g/m/sq.d)*			
	Distance	***Mean***	***Stdev***	***n***	***95% CL***
Midwater Canister	-15.00	17.11	4.29	9	2.80
	1.00	9.35	7.33	8	5.08
	10.00	3.52	4.95	8	3.43
	30.00	1.51	2.03	8	1.40
	50.00	0.70	0.74	7	0.55
	75.00	0.52	0.58	8	0.40
	125.00	0.14	0.03	3	0.03
	225.00	0.30	0.23	6	0.18

Figure 40: Organic waste material loading (g/m^2/day) at Young Passage farm site – Bottom Canister Data. Average inputs (lower table) shown graphically in spatial relationship with netcage system and with reference to background levels of sedimentation (green band is mean ± 95% CI). Red arrow indicates distance beyond which farm wastes can not be differentiated from background loading.



		Total Waste (g/m/sq.d)			
	Distance	Mean	Stdev	n	95% CL
Bottom Canister	-15.00	17.11	4.29	9	2.80
	1.00	13.89	6.28	8	4.35
	10.00	8.42	5.29	8	3.67
	30.00	3.03	1.78	8	1.23
	50.00	1.09	0.73	7	0.54
	75.00	0.82	0.70	8	0.49
	125.00	0.38	0.22	8	0.15
	225.00	0.34	0.22	6	0.17

The dispersion of wastes through the water column is summarized in Figure 41 for the Venture Point study site. Accumulation of quantifiable levels of this material, above that of background, could not be detected any further than 20 meters downstream of the cage system.

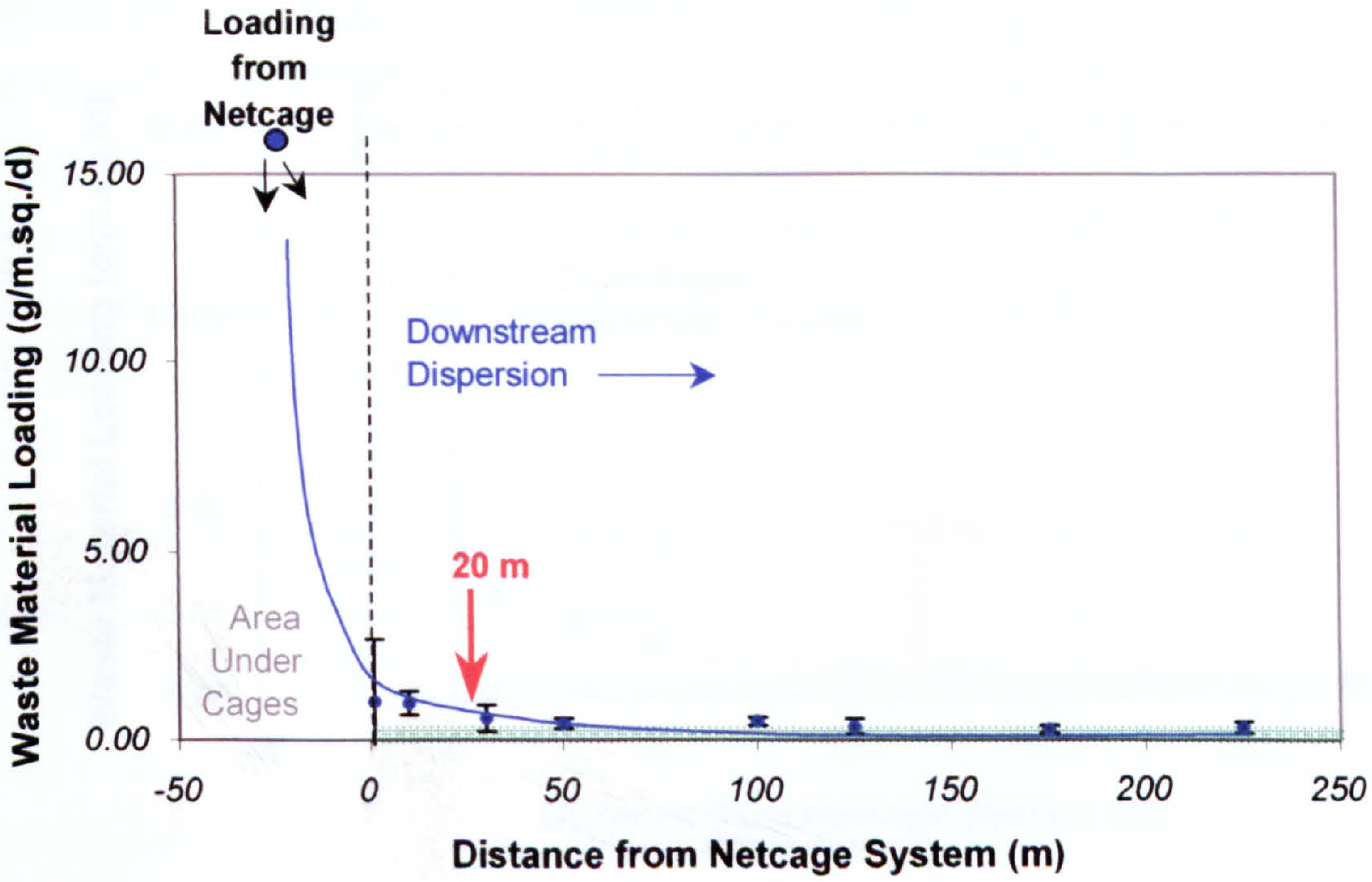
At the cage perimeter waste flux was estimated at 1.02 ± 0.32 $g/m^2/day$ (95% CL, n=8), only 6% of that measured leaving the cage system (Figure 41). The virtual loss of waste after leaving the system is further reflected in the remaining downstream stations, with only 0.98 ± 0.35 $g/m^2/day$ (95% CL, n=8) estimated at the 10-meter station and a decreasing flux at each subsequent station.

The bottom canisters suggest that some accumulation of waste material occurs at the Venture Pint site, despite the rapid dilution and dispersion processes that are apparent within the upper reaches of the water column. Figure 42 illustrates the extent of waste accumulation downstream of this study site, suggesting that measurable flux (related to the farm wastes) could not be detected beyond 115 meters of the cage system.

The rapid dispersion of this material is also reflected in these bottom data. At the farm perimeter, only 21.3% of the material (3.09 ± 0.98 $g/m^2/day$; 95% CL, n=7) discharged from the cage accumulated at the seafloor adjacent to the farm site. Furthermore, the waste material flux was <1.0 $g/m^2/day$ at approximately 75 meters downstream.

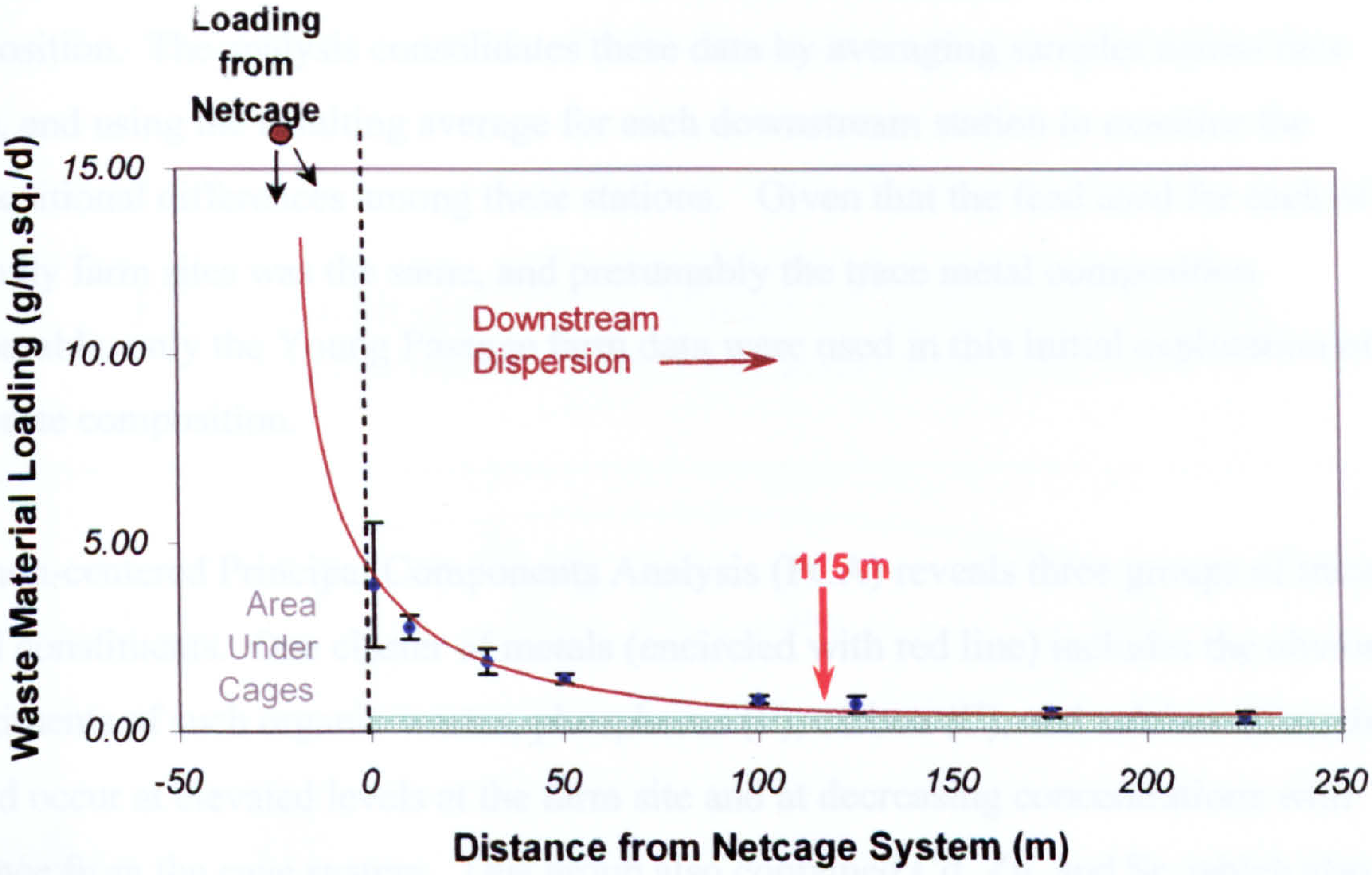
A very shallow dispersion curve, downstream of the cage system, implies a widespread distribution of this waste material. Although the loss of comparable levels of waste at the cage itself is apparent between the two study farm sites, at the Venture Point site the amount detected at the sediment water interface (and within the water column) is near the levels recorded for the natural, background sedimentation rates for this coastal area (0.386 ± 0.067 $g/m^2/day$; 95% CL, n=40).

Figure 41: Organic waste material loading (g/m^2/day) at Venture Point farm site – Midwater Canister Data. Average inputs (lower table) shown graphically in spatial relationship with netcage system and with reference to background levels of sedimentation (green band is mean ± 95% CI). Red arrow indicates distance beyond which farm wastes can not be differentiated from background loading.



		Total Waste (g/m/sq.d)			
	Distance	*Mean*	*Stdev*	*n*	*95% CL*
Midwater Canister	-15	18.31	3.15	14	1.65
	1	1.02	0.59	13	0.32
	10	0.98	0.51	8	0.35
	30	0.58	0.19	8	0.13
	50	0.44	0.16	7	0.12
	100	0.50	0.26	6	0.21
	125	0.36	0.09	2	0.12
	175	0.29	0.14	3	0.16
	225	0.32	0.15	7	0.11

Figure 42: Organic waste material loading (g/m^2/day) at Venture Point farm site – Bottom Canister Data. Average inputs (lower table) shown graphically in spatial relationship with netcage system and with reference to background levels of sedimentation (green band is mean + 95% CI). Red arrow indicates distance beyond which farm wastes can not be differentiated from background loading.



		Total Waste (g/m/sq.d)			
	Distance	***Mean***	***Stdev***	***n***	***95% CL***
Bottom Canister	-15	18.31	3.15	14	1.65
	1	3.90	1.23	7	0.91
	10	2.79	1.62	7	1.20
	30	1.89	0.46	8	0.32
	50	1.41	0.58	8	0.40
	100	0.86	0.39	6	0.32
	125	0.72	0.07	2	0.10
	175	0.50	0.22	7	0.16
	225	0.40	0.26	9	0.17

4.4.3 Chemical Constituents of Organic Wastes

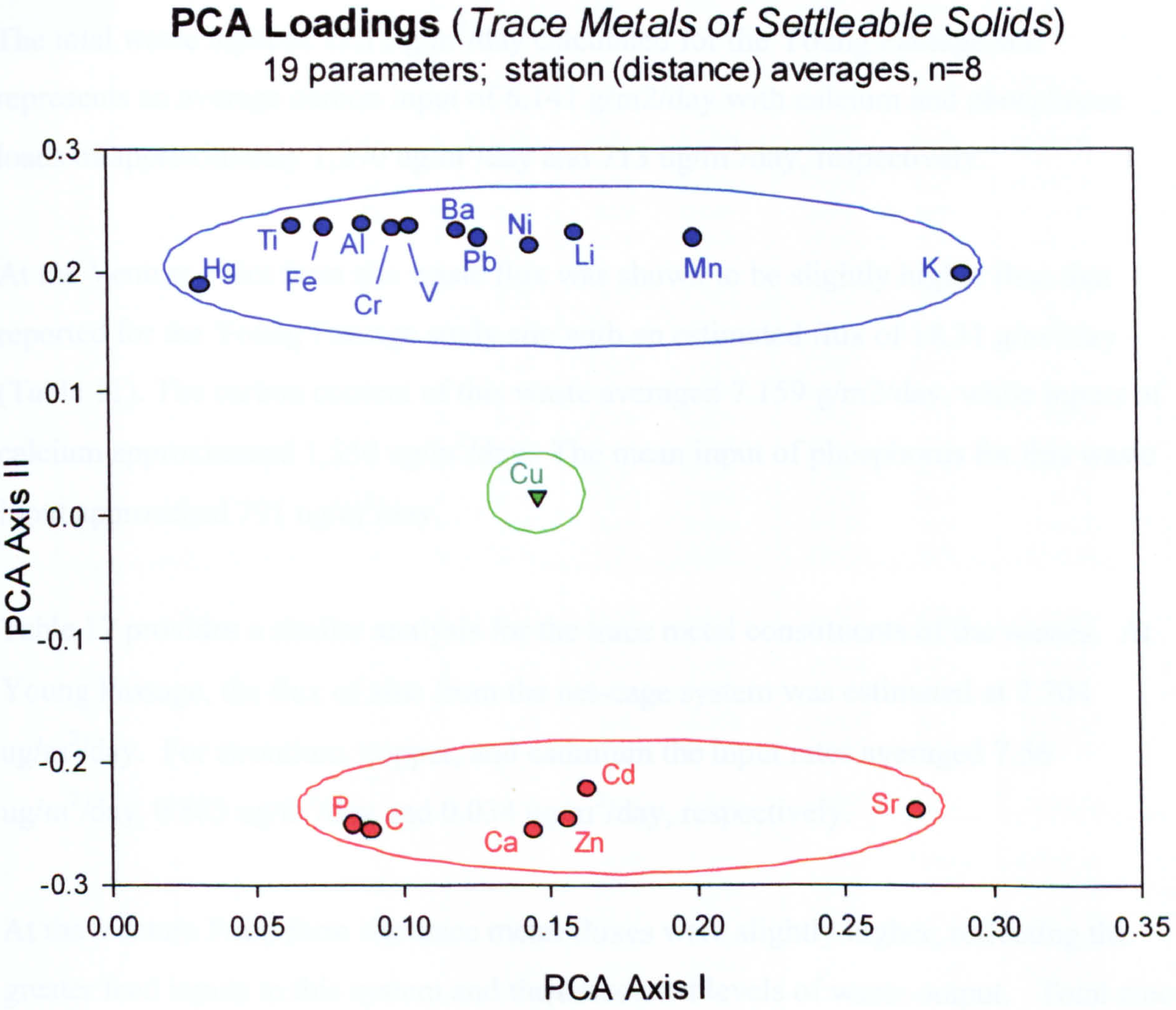
A chemical analysis of the waste samples acquired from the canister deployments included routine evaluation of trace metal composition of this waste, which comprised a scan of 19 specific constituents. Figure 43 uses all of the canister results from the Young Passage farm site database to classify samples according to trace metal composition. The analysis consolidates these data by averaging samples across time (n=8), and using the resulting average for each downstream station to examine the compositional differences among these stations. Given that the feed used for each of the study farm sites was the same, and presumably the trace metal composition comparable, only the Young Passage farm data were used in this initial exploration of the waste composition.

The non-centered Principal Components Analysis (PCA) reveals three groups of trace metal constituents. One cluster of metals (encircled with red line) includes the obvious constituents of such organic wastes, phosphorus (P), carbon (C), and calcium (Ca), that would occur at elevated levels at the farm site and at decreasing concentrations with distance from the cage system. This group also contained Cd, Zn, and Sr, which also occurred in slightly elevated concentrations in samples close to the farm, and thus assumed to be associated with farm waste inputs.

Copper (Cu) formed an isolated trace metal constituent in the Principal Components Analysis (Figure 43), given its use as a micronutrient of the feed and its likely short-term (and localized) flux from freshly treated farm nets (antifoulants). The remaining 12 trace metals formed a single grouping (blue circle), and encompassed variables such as potassium (K), manganese (Mn), lithium (L), nickel (Ni), barium (Ba), lead (Pb), vanadium (V), chromium (Cr), aluminum (Al), iron (Fe), tin (Ti), and mercury (Hg). These constituents were found in background sediments, collected furthest from the farm system, and there respective proportions in collected samples close to the farm decreased as the farm-derived waste component of the collection material increased. These metals did not appear to be associated with the waste flux that occurred from the farm itself.

Figure 43:

Principal Components Analysis (PCA) for trace metal constituents of settleable solids retained in waste collection canisters deployed at Young Passage farm site over duration of study. PCA of station/distance data (n=8) using averages (across time) for each of the 19 variables. 98.7% of variation accounted for in PCA axes I/II.



From these PCA results it was assumed that the potentially best trace metal "*indicators*" for the farm waste included carbon, phosphorus, calcium, zinc, selenium, and cadmium. Tables 3 and 4 present the concentrations for these constituents, as well as an estimate of their respective fluxes based on these concentration values and the total mass of organic waste measured for each station. Comparisons between mid-water canister fluxes and near-bottom loadings are provided for the Young Passage study site (dark print) and the Venture Point farm site (blue print) in each of these tables.

Table 11 reports the total waste flux for each farm site, as well as the concentrations and estimated fluxes for carbon, calcium and phosphorus components of the waste. The total waste input of 17.11 g/m^2/day calculated for the Young Passage site represents an average carbon input of 6.141 g/m2/day with calcium and phosphorus loads of approximately 1,390 ug/m^2/day and 713 ug/m^2/day, respectively.

At the Venture Point farm site waste flux was shown to be slightly higher than that reported for the Young Passage study site with an estimated flux of 18.31 g/m^2/day (Table 11). The carbon content of this waste averaged 7.159 g/m2/day, while inputs of calcium approximated 1,550 ug/m^2/day. The mean input of phosphorus for this waste input approached 791 ug/m^2/day.

Table 12 provides a similar analysis for the trace metal constituents of the wastes. At Young Passage, the flux of zinc from the net-cage system was estimated at 7.704 ug/m^2/day. For strontium, copper, and cadmium the input rates averaged 7.56 ug/m^2/day, 0.883 ug/m^2/day and 0.034 ug/m^2/day, respectively.

At the Venture Point farm site trace metal fluxes were slightly higher, reflecting the greater feed inputs to this system and the associated levels of waste output. Total zinc averaged 9.414 ug/m2/day, while inputs of strontium were estimated at 7.87 ug/m^2/day. The flux of copper was 0.974 ug/m^2/day and that for cadmium estimated at 0.029 ug/m^2/day for this farm site.

Table 11:

Total organic waste input from sediment canister data acquired from Young Passage and Venture Point farm sites, with flux estimates calculated for carbon, calcium and phosphorus using waste concentrations. Values based on average waste input determined from multiple canisters deployments (n=8). Flux as ug/m^2/d for Ca and P, and g/m^2/d for carbon. Table shows spatial trends (downstream stations) for midwater and bottom canister data.

			Total Waste (g/m/sq.d)				Carbon		T-Calcium		T-Phosphorus	
		Distance	Mean	Stdev	n	95% CL	conc. (%)	flux (g/m.sq./d)	conc. (ug/g)	flux (ug/m.sq./d)	conc. (ug/g)	flux (ug/m.sq./d)
VENTURE POINT SITE	Mid-Water	-15	18.31	3.15	14	1.65	39.10	7.159	84620	1549.3	43200.00	790.9
		1	1.02	0.59	13	0.32	21.57	0.219	72100	73.3	35042.86	35.6
		10	0.98	0.51	8	0.35	12.62	0.124	53540	52.4	20080.00	19.7
		30	0.58	0.19	8	0.13	10.15	0.059	44650	26.0	14985.00	8.7
		50	0.44	0.16	7	0.12	8.95	0.039	35600	15.7	10020.00	4.4
		100	0.50	0.26	6	0.21	6.83	0.035	36967	18.7	7146.67	3.6
		125	0.36	0.09	2	0.12	5.19	0.019	26500	9.5	3980.00	1.4
		175	0.29	0.14	3	0.16						
		225	0.32	0.15	7	0.11	3.95	0.013	28750	9.3	4072.50	1.3
	Bottom	-15	18.31	3.15	14	1.65	39.10	7.159	84620	1549.3	43200.00	790.9
		1	3.90	1.23	7	0.91	28.22	1.101	86880	338.9	41680.00	162.6
		10	2.79	1.62	7	1.20	30.95	0.864	74125	206.8	36425.00	101.6
		30	1.89	0.46	8	0.32	25.64	0.485	69620	131.8	32380.00	61.3
		50	1.41	0.58	8	0.40	23.08	0.326	54780	77.3	28420.00	40.1
		100	0.86	0.39	6	0.32	13.27	0.114	79667	68.2	17400.00	14.9
		125	0.72	0.07	2	0.10	10.85	0.078	48550	35.1	11455.00	8.3
		175	0.50	0.22	7	0.16	8.57	0.043	32220	16.2	6194.00	3.1
		225	0.40	0.26	9	0.17	5.77	0.023	33440	13.3	5002.00	2.0
YOUNG PASSAGE SITE	Mid-Water	-15	17.11	4.29	9	2.80	35.89	6.141	81257	1390.6	41642.86	712.7
		1	9.35	7.33	8	5.08	32.03	2.994	74800	699.0	32700.00	305.6
		10	3.52	4.95	8	3.43	26.78	0.943	66017	232.5	17566.67	61.9
		30	1.51	2.03	8	1.40	20.19	0.305	52386	79.2	17785.71	26.9
		50	0.70	0.74	7	0.55	15.52	0.109	30060	21.1	10430.00	7.3
		75	0.52	0.58	8	0.40	8.12	0.043	36015	18.9	8366.67	4.4
		125	0.14	0.03	3	0.03						
		225	0.30	0.23	6	0.18	6.21	0.019	29600	9.0	6081.43	1.8
	Bottom	-15	17.11	4.29	9	2.80	35.89	6.141	81257	1390.6	41642.86	712.7
		1	13.89	6.28	8	4.35	33.07	4.592	81017	1125.1	37583.33	521.9
		10	8.42	5.29	8	3.67	26.47	2.228	65283	549.7	31950.00	269.0
		30	3.03	1.78	8	1.23	24.91	0.755	50071	151.7	20342.86	61.6
		50	1.09	0.73	7	0.54	14.52	0.159	47420	51.8	14300.00	15.6
		75	0.82	0.70	8	0.49	9.81	0.081	45167	37.3	11391.67	9.4
		125	0.38	0.22	8	0.15	6.07	0.023	27650	10.5	7488.33	2.8
		225	0.34	0.22	6	0.17	2.60	0.009	21257	7.2	4388.57	1.5
Background Levels:			0.37	0.22	40	0.07	4.65	0.017	30329	11.2	4917	1.8

Table 12:

Sediment canister data acquired from Young Passage and Venture Point farm sites, showing flux estimates (ug/m^2/d) calculated for total zinc, cadmium, strontium and copper using waste concentrations of these trace metal constituents and proportion of total organic waste input (see Table 11). Values based on average waste input determined from multiple canisters deployments (n=8). Table shows spatial trends (downstream stations) for midwater and bottom canister data.

			T-Zinc		T-Cadmium		T-Strontium		T-Copper	
		Distance	conc. (ug/g)	flux (ug/m.sq./d)	conc. (ug/g)	flux (ug/m.sq./d)	conc. (ug/g)	flux (ug/m.sq./d)	conc. (ug/g)	flux (ug/m.sq./d)
VENTURE POINT SITE	Mid-Water	-15	514.20	9.414	1.58	0.029	430.00	7.87	53.20	0.974
		1	476.14	0.484	1.73	0.002	378.43	0.38	58.86	0.060
		10	338.20	0.331	1.54	0.002	327.60	0.32	59.00	0.058
		30	302.25	0.176	1.28	0.001	303.25	0.18	63.75	0.037
		50	366.00	0.161	1.48	0.001	270.00	0.12	74.50	0.033
		100	171.33	0.087	1.07	0.001	292.67	0.15	56.00	0.028
		125	98.00	0.035	1.00	0.000	250.33	0.09	38.33	0.014
		175								
		225	140.50	0.045	0.80	0.000	312.75	0.10	44.25	0.014
	Bottom	-15	514.20	9.414	1.58	0.029	430.00	7.87	53.20	0.974
		1	521.60	2.035	2.40	0.009	439.80	1.72	74.00	0.289
		10	442.00	1.233	2.15	0.006	394.25	1.10	40.50	0.113
		30	400.60	0.758	2.12	0.004	384.60	0.73	54.00	0.102
		50	367.20	0.518	1.66	0.002	340.20	0.48	59.40	0.084
		100	230.67	0.198	1.30	0.001	423.33	0.36	96.33	0.082
		125	203.00	0.147	1.30	0.001	318.50	0.23	43.50	0.031
		175	136.80	0.069	1.24	0.001	238.00	0.12	56.40	0.028
		225	117.80	0.047	0.88	0.000	250.00	0.10	52.80	0.021
YOUNG PASSAGE SITE	Mid-Water	-15	450.14	7.704	2.00	0.034	442.00	7.56	51.57	0.883
		1	366.00	3.420	2.25	0.021	486.67	4.55	110.33	1.031
		10	258.50	0.910	2.35	0.008	484.17	1.71	253.17	0.892
		30	241.86	0.366	2.16	0.003	406.71	0.62	250.29	0.379
		50	191.40	0.134	1.74	0.001	485.40	0.34	180.60	0.127
		75	187.67	0.098	1.12	0.001	358.50	0.19	130.67	0.069
		125								
		225	168.86	0.051	0.57	0.000	336.86	0.10	87.57	0.027
	Bottom	-15	450.14	7.704	2.00	0.034	442.00	7.56	51.57	0.883
		1	426.83	5.927	1.93	0.027	426.17	5.92	66.83	0.928
		10	387.83	3.266	2.03	0.017	360.83	3.04	48.67	0.410
		30	259.57	0.787	1.81	0.005	361.57	1.10	114.86	0.348
		50	287.60	0.314	1.54	0.002	389.00	0.42	143.20	0.156
		75	212.83	0.176	1.83	0.002	393.67	0.32	123.33	0.102
		125	269.00	0.102	0.53	0.000	236.00	0.09	109.67	0.041
		225	111.24	0.038	0.36	0.000	213.86	0.07	61.86	0.021
Background Levels:			135.80	0.050	0.60	0.000	279.52	0.10	60.48	0.022

series of graphs that illustrate dispersion of the waste downstream of the study sites in terms of these chemical "signals". Plotted values represent average constituent concentration, using data from each of the canister deployment periods, with 95% confidence limits as a measure of variability (error bars).

Figure 44 and Figure 45 present these data for the Venture Point mid-water collection region and the near-bottom collection region, respectively. The red vertical line indicates the distance at which no significant difference in total waste was determined. The green horizontal line show average reference levels with 95% confidence limits.

For the mid-water column data the constituent concentrations (Figure 44) appeared to follow a similar dispersion pattern to that indicated by the total waste flux (shown in Figure 41). Levels quickly approached reference conditions, estimated at 20 meters downstream, and all of these data represented a very small proportion of the total flux (and concentration) estimated at the net-cage itself. Exceptions to this trend were noted in the phosphorus, zinc and cadmium components that suggested a slightly greater dispersion, estimated at between 40 and 75 meters for the former two and as far as 100 meters for the cadmium concentrations. All concentrations and estimated fluxes at these distances were very low, and in the case of cadmium close to the level of detection for the analytical methodologies (which varied as a result of sample volume submitted).

For the near-bottom canister data (Figure 45) the concentrations for the waste constituents again revealed a similar dispersion pattern to that reported for the total waste flux (shown in Figure 42). A significant downstream signal in these key waste constituents was shown to occur to a distance of 115 meters for each of these parameters, with levels at or below that of the background or reference conditions. An exception was again noted with total cadmium concentrations, which appeared to level out before the 115-meter distance yet continued to report concentrations in excess of the reference well beyond this point. The very low concentrations measured for this parameter beyond 75 meters may reflect the difference in this dispersion result.

Figure 44:

Chemical *signal* of organic waste material collected downstream of Venture Point farm site: **mid-water canisters**. Concentration of the key elements is shown in relation to reference levels (green line), and with respect to distance from net-cage perimeter. The in-cage station is shown within the gray-shaded region at -15 meters. All values are means with 95% C.L.'s.

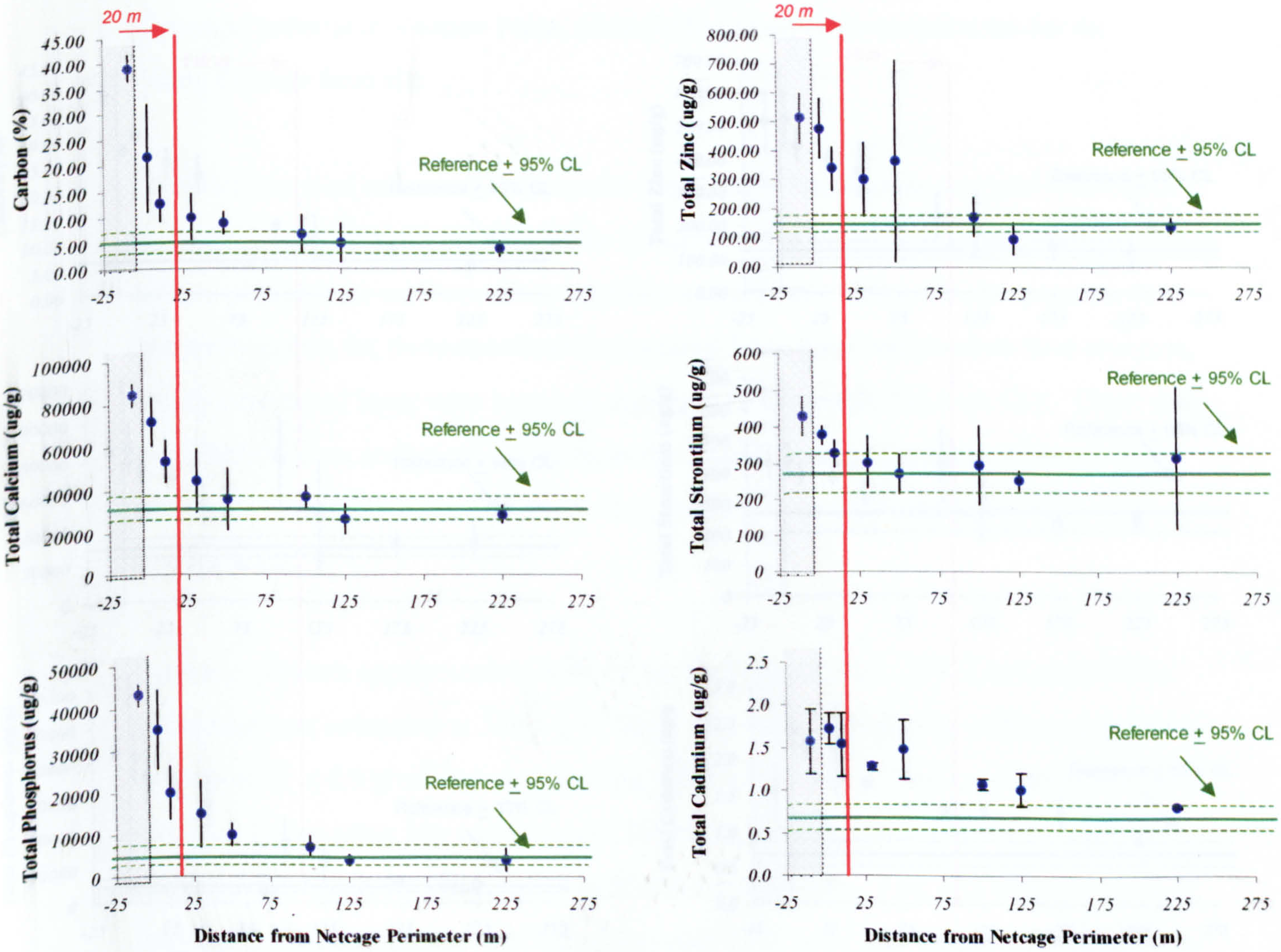
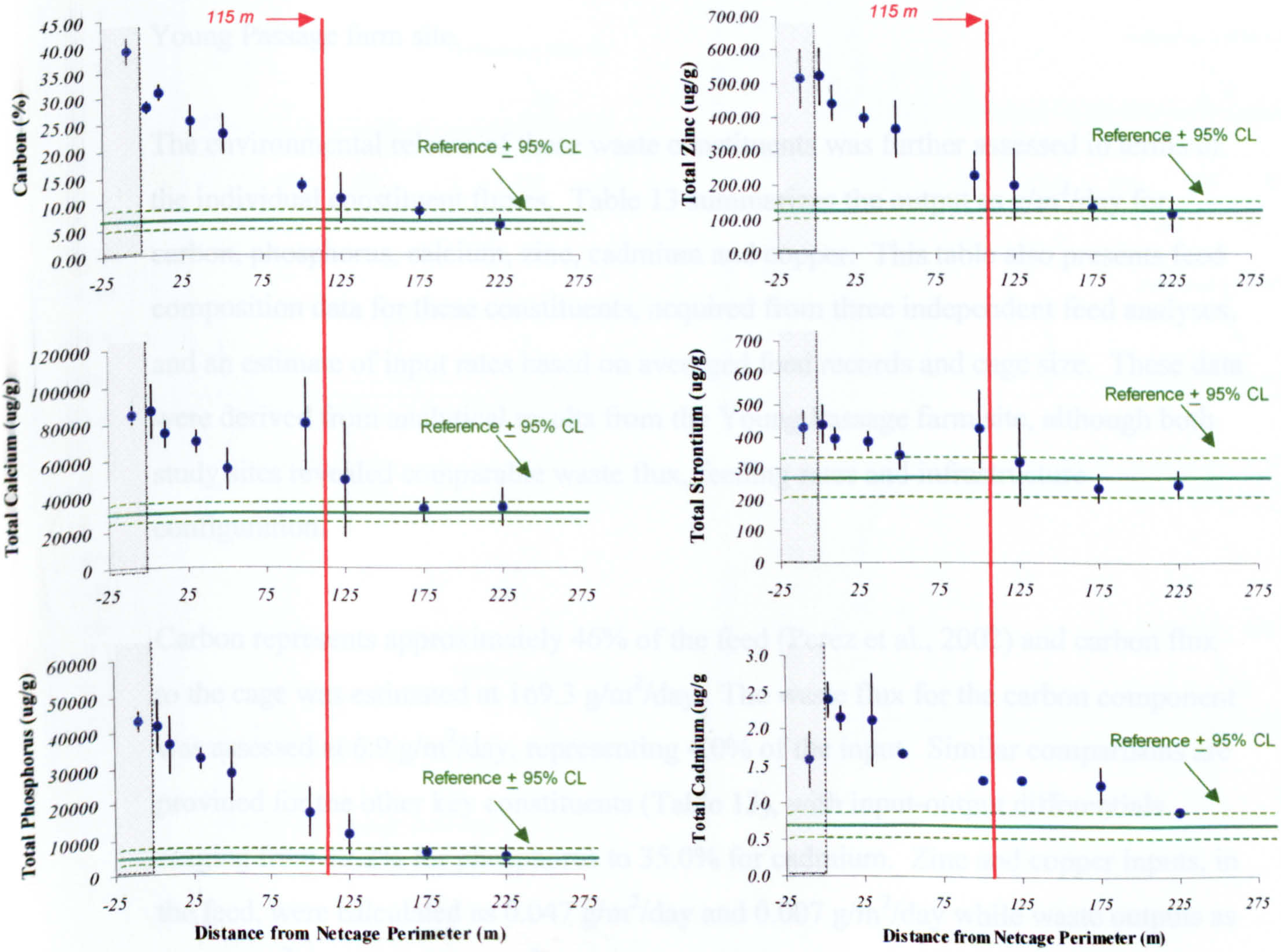


Figure 45:

Chemical signal of organic waste material collected downstream of Venture Point farm site: ***bottom canisters***. Concentration of the key elements is shown in relation to reference levels (green line), and with respect to distance from net-cage perimeter. The in-cage station is shown within the gray-shaded region at -15 meters. All values are means with 95% C.L.'s.



At the Young Passage farm site the dispersion curves for the total waste, shown previously in Figures 39 and 40 for the mid-water and near-bottom canisters, were closely replicated in the spatial assessment of the key waste constituents (Figures 46 and 47). The mid-water dispersion of these key waste components extended to 60 meters downstream (Figure 46), while the near-bottom distribution of these contaminants extended to 100 meters (Figure 47).

Both sets of graphs (Figures 46, 47) suggest that the trace metal "signal" closely follows that of the total waste flux at the Young Passage farm site. The cadmium concentrations, which appeared more widespread and contradictory of other component dispersion patterns at Venture Point, were similar to all other constituents for the Young Passage farm site.

The environmental release of these waste constituents was further assessed in terms of the individual constituent fluxes. Table 13 summarizes the output as g/m^2/day for carbon, phosphorus, calcium, zinc, cadmium and copper. This table also presents feed composition data for these constituents, acquired from three independent feed analyses, and an estimate of input rates based on averaged feed records and cage size. These data were derived from analytical results from the Young Passage farm site, although both study sites revealed comparable waste flux, feeding rates and infrastructure configuration.

Carbon represents approximately 46% of the feed (Perez et al., 2002) and carbon flux to the cage was estimated at 169.3 g/m^2/day. The waste flux for the carbon component was assessed at 6.9 g/m^2/day, representing 4.0% of the input. Similar comparisons are provided for the other key constituents (Table 13), with input-output differentials ranging from 12.2% for phosphorus to 35.0% for cadmium. Zinc and copper inputs, in the feed, were calculated as 0.047 g/m^2/day and 0.007 g/m^2/day while waste outputs as 0.009 g/m^2/day and 0.001 g/m^2/day, respectively. The proportion of the feed input lost as solid waste was comparable between these two metals, with percentages of 18.0 for zinc and 20.6 for copper.

Figure 46:

Chemical *signal* of organic waste material collected downstream of Young Passage farm site: **mid-water canisters**. Concentration of the key elements is shown in relation to reference levels (green line), and with respect to distance from net-cage perimeter. The in-cage station is shown within the gray-shaded region at -15 meters. All values are means with 95% C.L.'s.

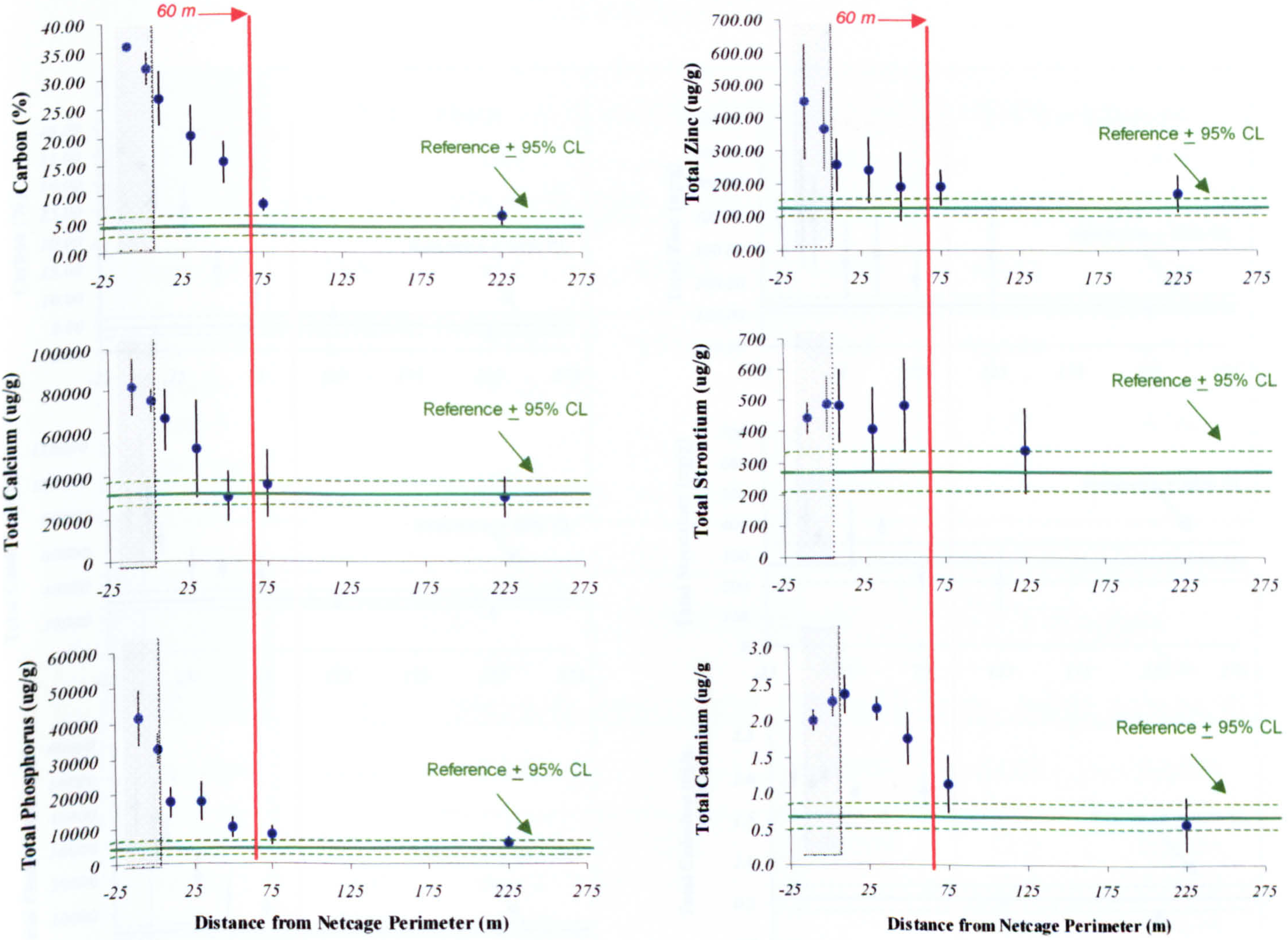


Figure 47:

Chemical signal of organic waste material collected downstream of Young Passage farm site: ***bottom canisters***. Concentration of the key elements is shown in relation to reference levels (green line), and with respect to distance from net-cage perimeter. The in-cage station is shown within the gray-shaded region at -15 meters. All values are means with 95% C.L.'s.

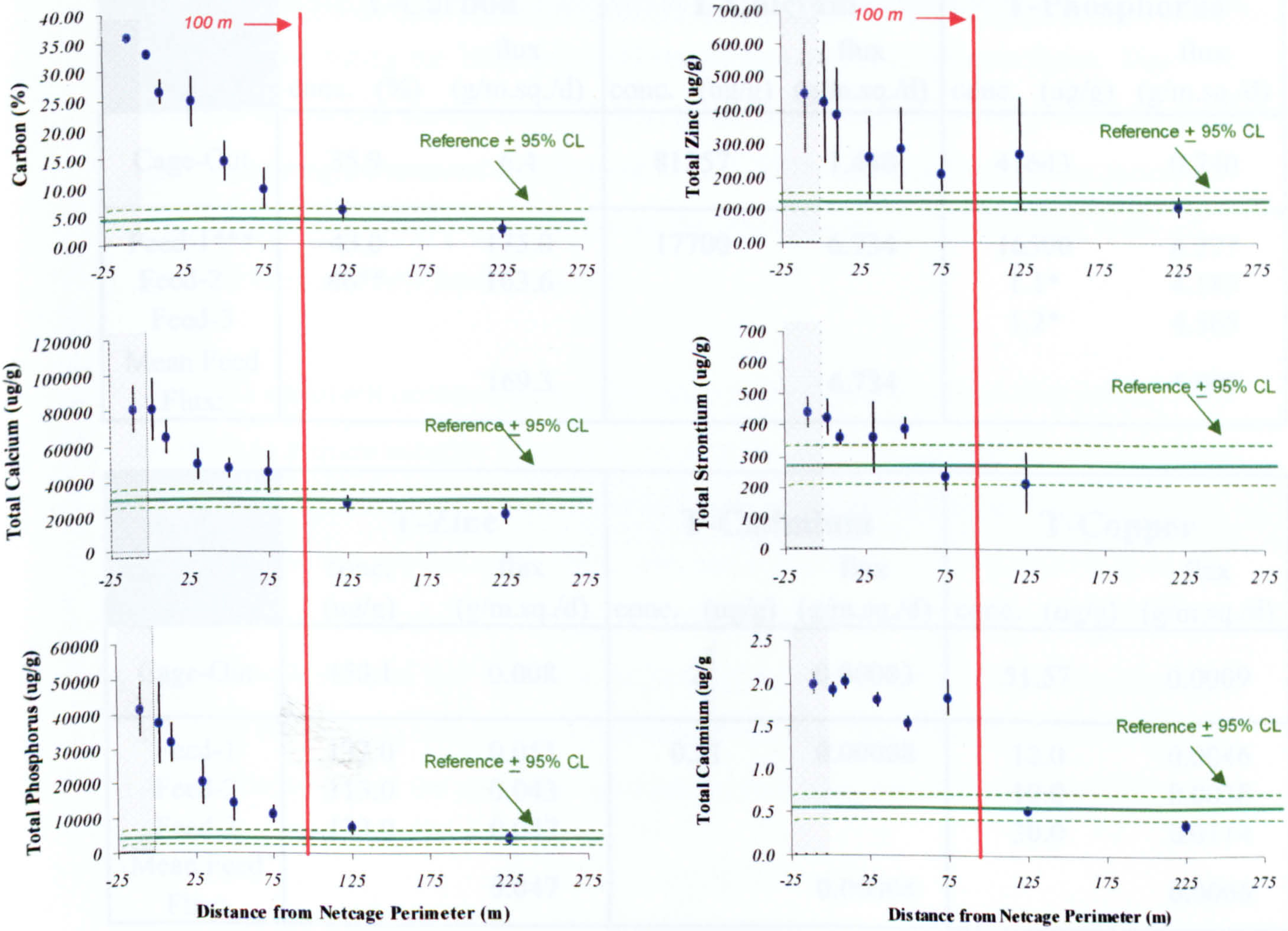


Table 13

Concentrations and estimated fluxes for key constituents of fish feed delivered to study sites. Concentrations were acquired from analytical results (this study, Feed-1) as well as from feed analyses conducted on samples by a feed supplier (Feed-2, Feed-3). Estimates of waste output provided as a comparison for inputs. Assumptions used to calculate fluxes shown at bottom of table.

	T-Carbon		**T-Calcium**		**T-Phosphorus**	
	conc. (%)	flux (g/m.sq./d)	conc. (ug/g)	flux (g/m.sq./d)	conc. (ug/g)	flux (g/m.sq./d)
Cage-Out	35.9	6.4	81257	1.440	41643	0.740
Feed-1***	43.0	175.0	17700	6.734	16500	6.277
Feed-2	46**	163.6			1.1*	4.185
Feed-3					1.2*	4.565
Mean Feed Flux:		169.3		6.734		5.009

	T-Zinc		**T-Cadmium**		**T-Copper**	
	conc. (ug/g)	flux (g/m.sq./d)	conc. (ug/g)	flux (g/m.sq./d)	conc. (ug/g)	flux (g/m.sq./d)
Cage-Out	450.1	0.008	2	0.00003	51.57	0.0009
Feed-1	133.0	0.051	0.21	0.00008	12.0	0.0046
Feed-2	113.0	0.043			10.0	0.0038
Feed-3	123.0	0.047			30.0	0.0114
Mean Feed Flux:		0.047		0.00008		0.0066

* *values as percent of feed*

** *estimate from Perz et al., 2002*

*** *analytical results acquired from this study*

4.4.4 Chemotherapeutic Treatment Effects

The application of chemotherapeutics occurred at each of the study sites as required of the respective production cycles, and any resulting environmental sampling was therefore completed opportunistically (once informed of the need for such treatments). The analytical results from these samples are provided for each of these treatments in the following subsections.

Emamectin benzoate (SLICE ^{TM}) Treatments

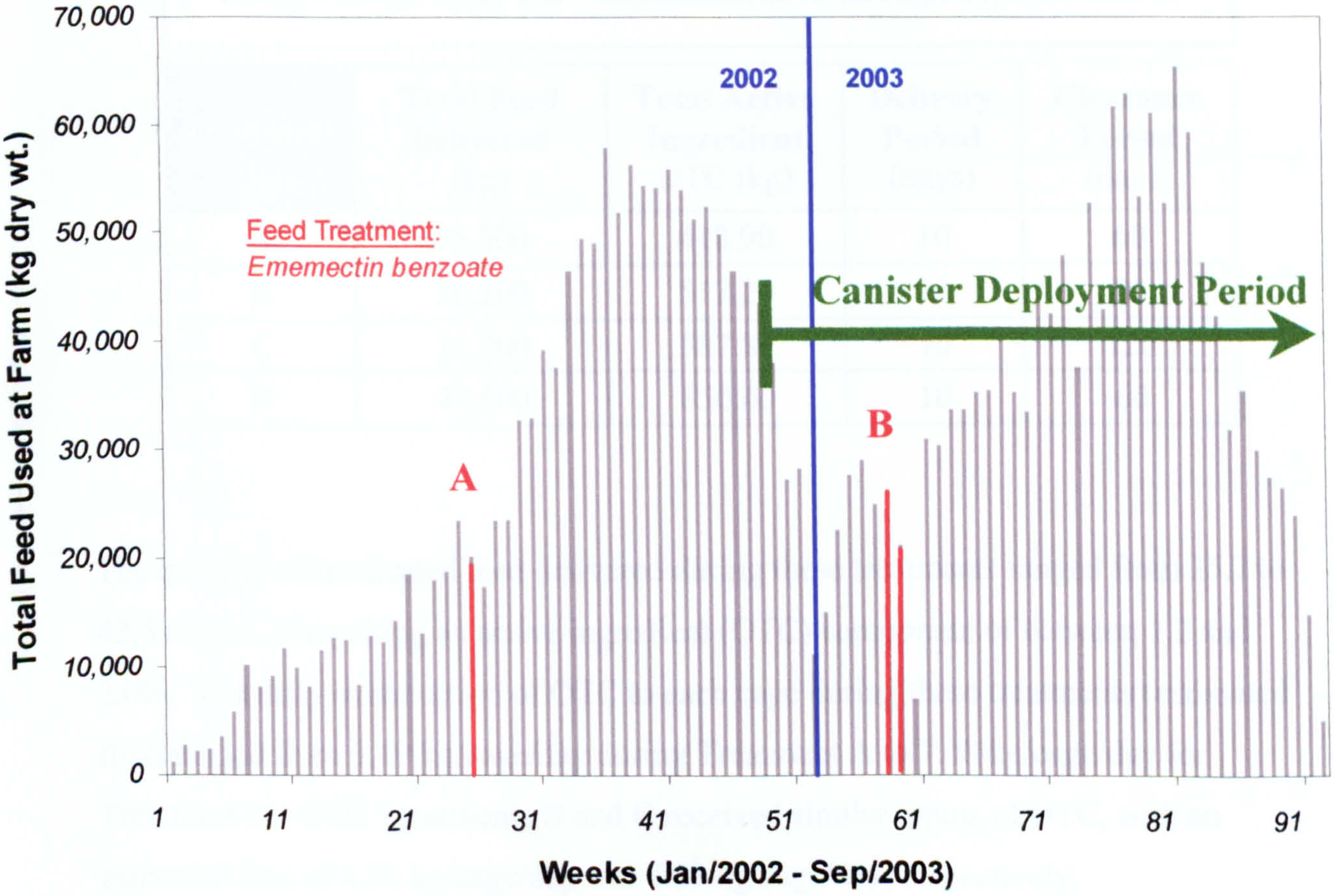
Levels of sea lice (*Caligus* sp.) increased to levels that warranted pesticide treatment on two occasions during the 2002 and 2003 production cycles at Venture Point. Figure 48 indicates the periods during which SLICE (Emamectin benzoate) was prescribed for the farm, indicating a treatment requirement in mid-2002 (half way through the production cycle) and in early 2003 just as the last of the 2002 harvest fish were being removed from the farm and new smolts entered.

The 2003 treatment occurred during the waste collection component of this study, and an effort to acquire samples from each canister that would document dispersion of this pesticide compound downstream of the farm site was made following this treatment. The pesticide was applied to the entire site, and comprised a total of 34,310 kg of treated feed with a prescribed dosage of 1.0%. Thus, the total amount of SLICE released with the feed was 343.1 kg, or 28.6 kg/cage, or an active ingredient loading (0.2% of the premix) of 5.72 g of emamectin benzoate. As this treatment was applied over a 10-day period the pesticide input rate to each cage was estimated at 0.572 g/cage/day.

A single set of sediment collection samples was analysed for SLICE residues post-treatment. A period of 1 week was allowed to pass following delivery of treated feed at the farm site prior to canister removal (collection period = 26 days). Despite a period of waste collection that spanned the feed delivery and a short period post-treatment, no emamectin benzoate residues (< 0.5 ug/g) were detected in the submitted samples.

Figure 48:

Weekly feed input (kg) to Venture Point farm site showing periods (red bars) during which feed containing sea lice treatment pesticide, *Ememectin benzoate* (Slice $_{TM}$), was delivered. Treatment A occurred over a 1-week period with 24,225 kg of medicated feed delivered. Treatment B extended over a 10-day period and comprised 34,310 kg of medicated feed. Period over which sediment canisters were deployed shown in green.



Oxytetracycline (OTC) Treatments

Oxytetracycline (OTC) treatments were applied at the Young Passage farm site on four occasions over the period of this research project. Table 14 summarizes the treatment specifications for each of these feed deliveries at the Young Passage farm site. The medication was milled with the feed and delivered to all fish retained at the site over treatment periods of 10 days. The predicted clearance periods associated with these treatments ranged from 60 to 120 days.

Table 14

Specifications for Oxytetracycline (OTC) treatments prescribed at the Young Passage study site – September/2002 through September/2003

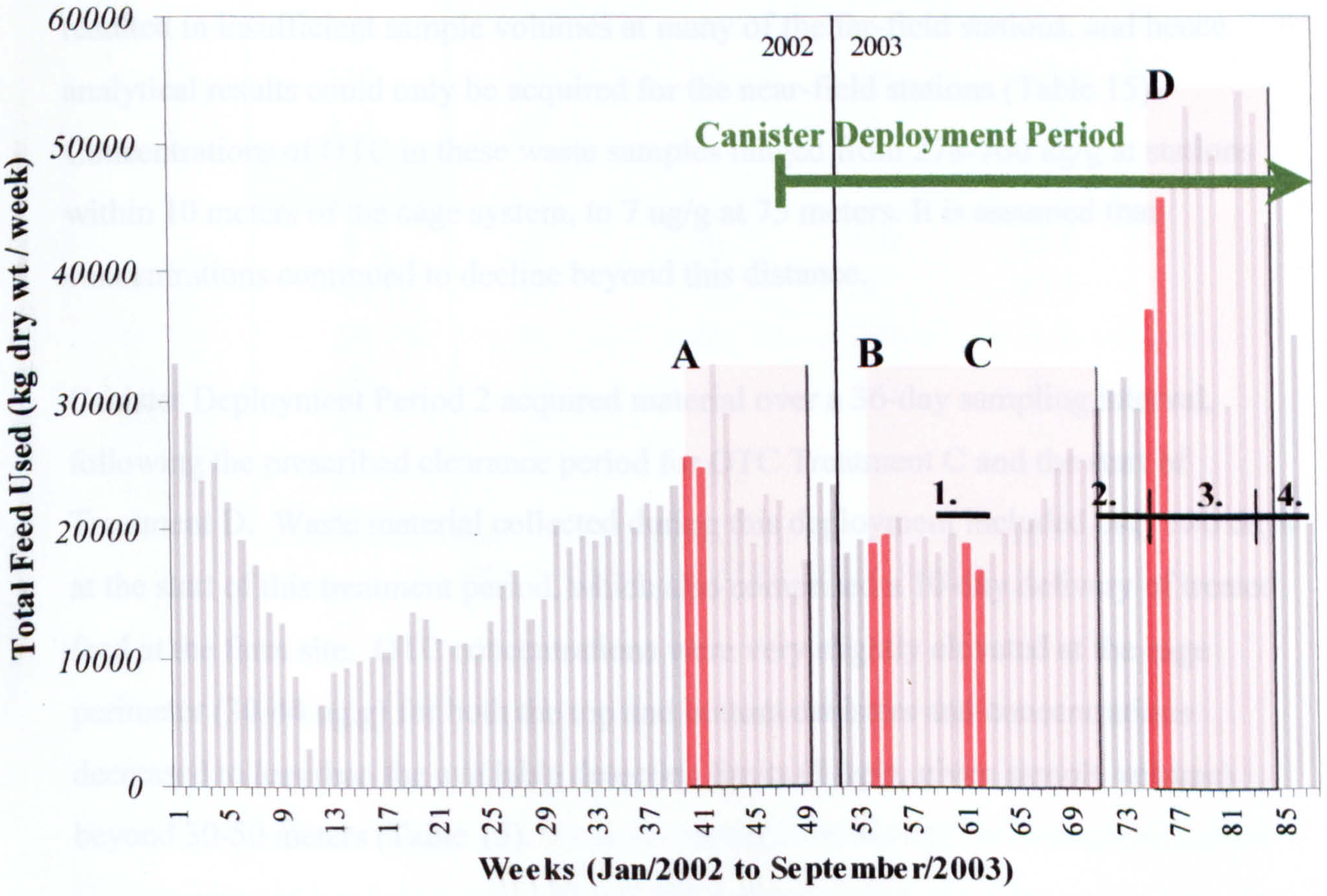
Treatment	**Total Feed Delivered** (kg)	**Total Active Ingredient** OTC (kg)	**Delivery Period** (days)	**Clearance Period** (days)
A	35,500	418.90	10	60
B	36,200	517.29	10	90
C	35,200	587.84	10	120
D	42,500	850.00	10	60

The quantity of medicated feed delivered during these treatments ranged from 35.2 to 42.5 tonnes, comprising an active ingredient (OTC) component of between 1.2 and 2.0%. The daily introduction of OTC to each cage during these treatments (estimated flux) ranged from 3.49 kg/cage/day during Treatment A to 7.08 kg/cage/day for Treatment D. OTC Treatments B and C received similar inputs of OTC, with an estimated flux of 4.31 kg/cage/day and 4.89 kg/cage/day, respectively.

Figure 49 presents the medicated (OTC) feed inputs (red bars) for the Young Passage farm site in relation to the weekly levels of feed delivered at the site (gray bars). The clearance period predicted for each treatment is portrayed using the light red shading, indicating considerable overlap between treatments B and C.

Figure 49:

Weekly feed input (kg) to Young Passage farm site (gray bars) showing oxytetracycline (OTC) treatment (red bars) and prescribed withdrawal periods (light red shaded areas). Each of the four required treatments (A-D) was delivered with medicated feed (milled) over a 10-day feeding period. Tonnage fed out to the fish over these treatments varied from 35 to 42 tonnes. Duration of canister deployments indicated with green arrow.



The duration of the canister deployments for this study are shown with the green line/arrow on Figure 49. Waste material collection spanned OTC Treatments B,C, and D, although residue analyses were completed for deployments that occurred over the medicated feed inputs, and residual clearance periods, associated with Treatments C and D. The canister deployments that were used for these OTC residue analyses are indicated numerically (1-4) on this figure.

Waste recovery from Canister Deployment 1 (26 days) was initiated prior to OTC Treatment C (Figure 49), extended through the treatment period (10 days) and was terminated a few days into the clearance period. The initial period of the canister deployment also coincided with the clearance period associated with Treatment B, which was completed one month earlier. The relatively short waste recovery period resulted in insufficient sample volumes at many of the far-field stations, and hence analytical results could only be acquired for the near-field stations (Table 15). Concentrations of OTC in these waste samples ranged from 278-760 ug/g at stations within 10 meters of the cage system, to 7 ug/g at 75 meters. It is assumed that concentrations continued to decline beyond this distance.

Canister Deployment Period 2 acquired material over a 36-day sampling interval, following the prescribed clearance period for OTC Treatment C and the start of Treatment D. Waste material collected during this deployment included only two days at the start of this treatment period, which also comprised a 10-day delivery of treated feed at the farm site. OTC concentrations were very slightly elevated at the cage perimeter (30-44 ug.g) for both the top and bottom canisters and concentrations decreased to less than the available detection limits (1 ug/g, given sample volume) beyond 30-50 meters (Table 15).

Canister Deployment 3 encompassed the entire medicated feed delivery and initial clearance periods for OTC Treatment D. Sample collection occurred contiguously with that of Canister Deployment 2, although the sampling interval comprised 57 days and included 8 days of the treated feed delivery and most of the prescribed clearance period.

Table 15

Concentration of oxytetracycline (OTC) in waste material collected in Canister Deployments 1 to 4 over Treatments C and D (Figure 49), including their associated clearance periods. Values are ug/g OTC, and are presented for mid-water and near-bottom canisters at stations established downstream of the Young Passage study site.

	Oxytetracycline Concentration (ug/g)			
TOP	**1**	**2**	**3**	**4**
-15	276	7	5	8.4
0	334	44	150	11
10	790	13	254	2.1
30	133	4	i/s	0.8
50	59	1	3	2.2
75	7	1	1	1.4
125	i/s	i/s	i/s	i/s
225	i/s	1	1	1.3
BOTTOM				
0	287	30	178	10.6
10	95	15	97	47.5
30	i/s	12	228	1.2
50	i/s	1	131	8.1
75	i/s	1	39	0.5
125	i/s	1	1	5.5
225	i/s	1	i/s	1.5

i/s = insufficient sample quantity for analysis

Table 15 indicates that OTC concentrations reported for this deployment showed a similar spatial trend to that of the previous canister deployment, although levels were higher by an order of magnitude. Stations within 50 meters of the netcage contained OTC residues at levels between 130 and 255 ug/g of collected waste.

Collecting waste material for 34 days, Canister Deployment 4 occurred entirely post-treatment and was initiated near the end of the prescribed clearance period for Treatment D. OTC concentrations were significantly less than those reported in the previous deployment, with a maximum of 47.5 ug/g at 10 meters downstream of the farm site within the bottom samples.

Figure 50 uses these OTC concentration data to examine the spatial and temporal trends in antibiotic residue dispersion. These data are contoured to show changes in downstream concentration patterns during and following the treatment of fish with OTC on two occasions during 2003 (February and June). The light yellow contour indicates a concentration of <1.0 ug/g, and represents the limit of detection used in processing these samples (governed in part by available sample material). The blue background represents an area that could not adequately be contoured given the combination of missing and <1.0 ug/g data values (blank regions). However, these patterns jointly suggest that little downstream OTC dispersion, associated with the settleable waste solids, occurred beyond 75 meters within the mid-water and 125 meters in the near-bottom.

Figure 50 also illustrates a clear peak in OTC concentration associated with each of the two treatments. The extrapolated information indicated that the water column concentrations decreased to background levels between treatments and that the water column clearance period ranged from 30-45 days from the onset of the therapeutic treatment.

To further explore the dispersion trends in OTC associated with the organic wastes originating from the study farms, canister concentrations were converted to flux values that were standardized according to the days exposed to treatment and post-treatment, or prescribed clearance periods (= proportion of canister deployment period). These fluxes were averaged across the four deployment periods for each downstream sampling station and presented as an average flux (± 95% confidence limits) for the mid-water and near-bottom regions assessed in this study component.

Figure 51 reveals the spatial trends observed in these OTC flux data. A negative exponential relationship was apparent in the flux-distance data for both the mid-water (upper graph) and near-bottom (lower graph). These curves ($r^2 = 0.756$ and 0.966) were similar in slope (decay) and different only in y-intercept (cage edge), which indicated an input of 134.3 ug/m^2/d for the mid-water and 1,012.5 ug/m^2/d for the near-bottom region.

Figure 50:

Temporal and spatial pattern of OTC dispersion at the Young Passage farm site. Mid-water and near-bottom waste collection data from 4 canister deployments (Figure 49) are presented independently. Duration of OTC Treatments C and D are indicated with horizontal line. Contoured concentration data (ug/g) extrapolate trends between sampling periods. Light yellow contour indicates concentrations at or below detection limit of 1.0 ug/g.

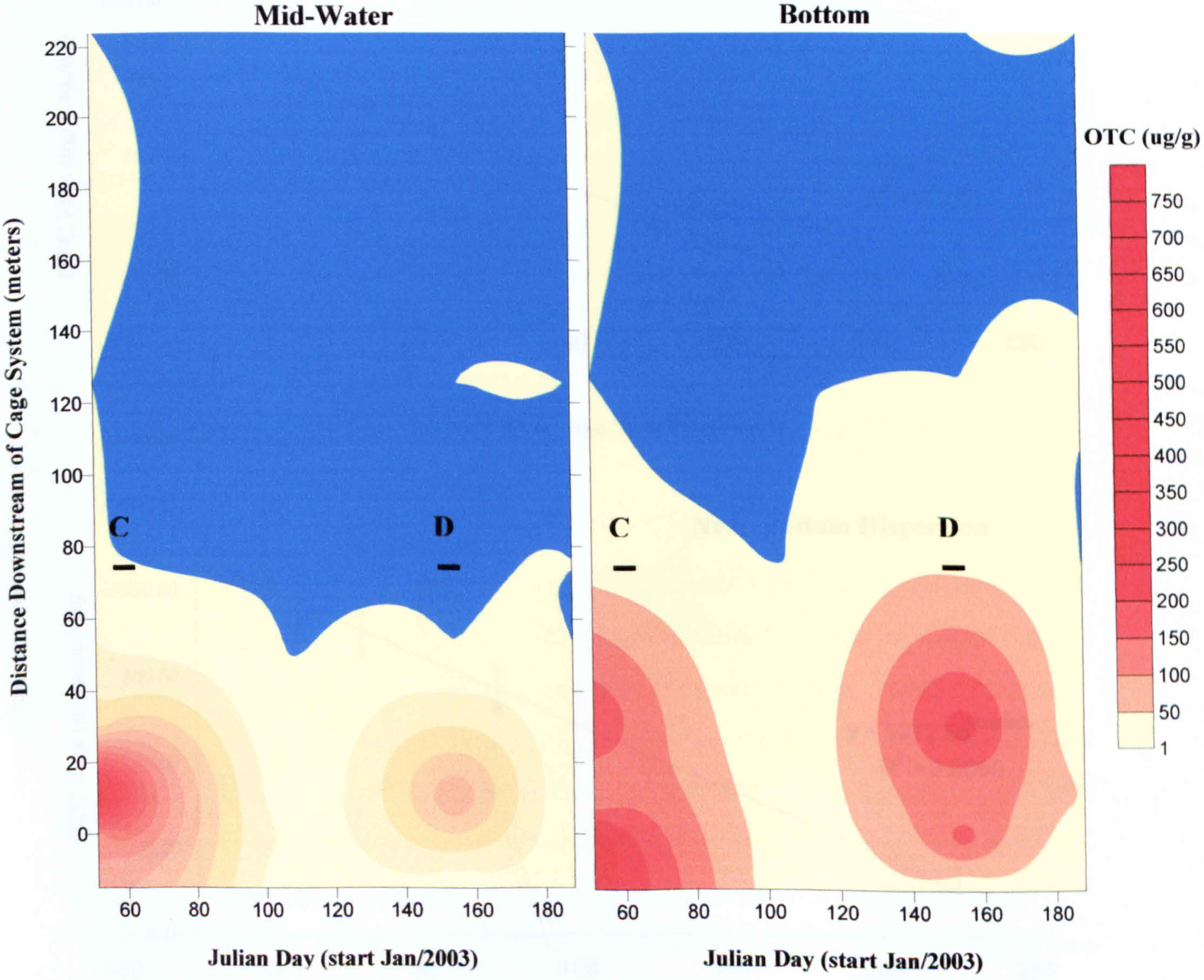
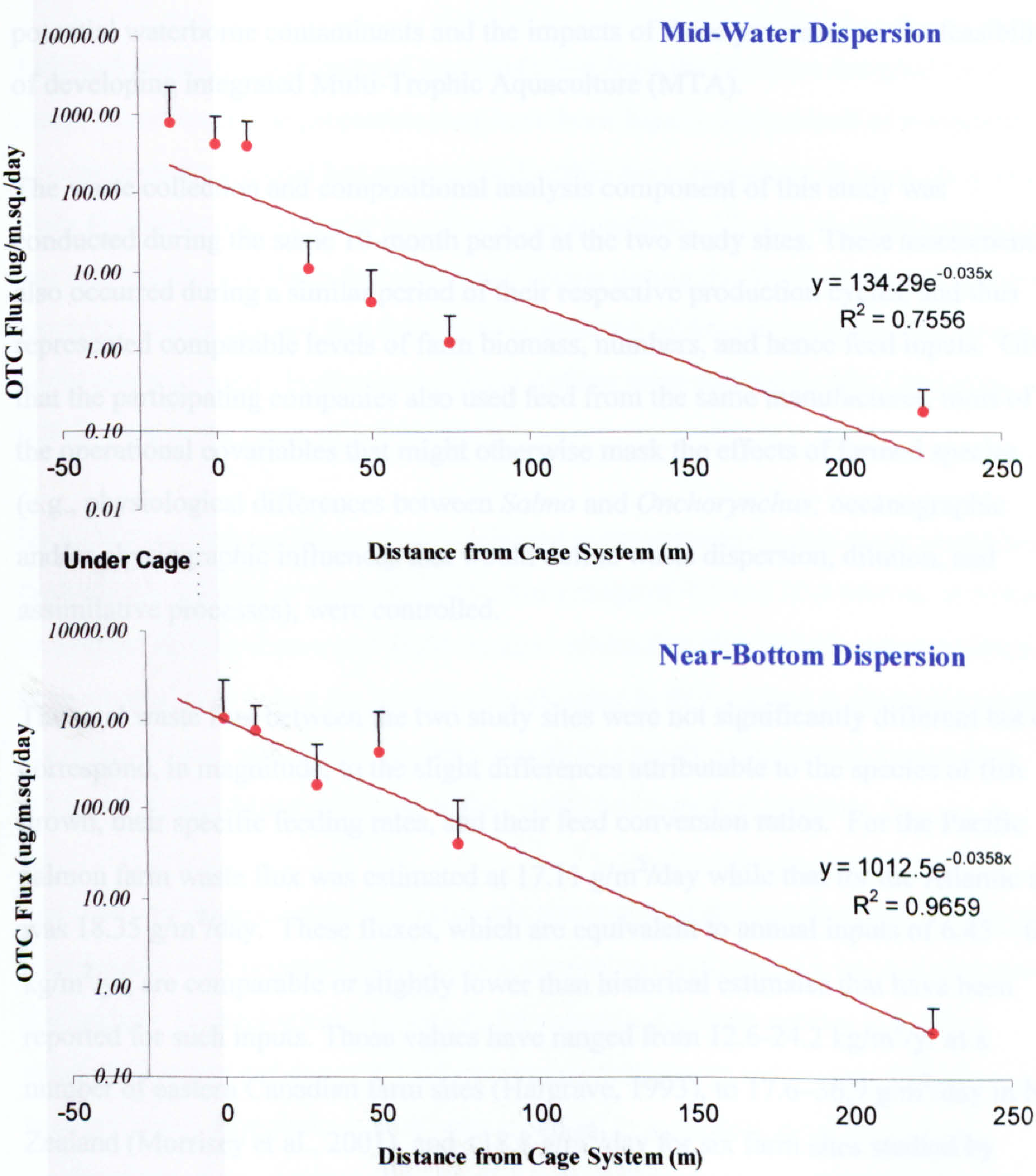


Figure 51:

Average flux of OTC residue in downstream organic waste samples collected at the Young Passage farm site over two therapeutic treatment and clearance periods (2003). Estimated flux presented as ug/m.sq./day of OTC detected in waste material, using total organic waste biomass to calculate values (standardized to OTC exposure period). Upper graph shows flux at each mid-water station and lower graph presents flux for near-bottom sampling stations. Variance (error bars) presented as 95% confidence limits (n=4).



4.5 Discussion

The results of this study component have provided valuable insight into waste material flux from marine finfish aquaculture facilities, and the physical processes by which the chemical constituents of these wastes become bioavailable to non-target species (such as shellfish) in the receiving environment. The use of two significantly different farm sites, in terms of their oceanographic and physiographic characteristics (discussed in Chapter 3), support a further assessment of the dilution and dispersion effects on these potential waterborne contaminants and the impacts of these processes on the feasibility of developing integrated Multi-Trophic Aquaculture (MTA).

The waste collection and compositional analysis component of this study was conducted during the same 10-month period at the two study sites. These assessments also occurred during a similar period of their respective production cycles, and thus represented comparable levels of farm biomass, numbers, and hence feed inputs. Given that the participating companies also used feed from the same manufacturer, most of the operational covariables that might otherwise mask the effects of farmed species (e.g., physiological differences between *Salmo* and *Onchorynchus;* oceanographic and/or physiographic influences that would define waste dispersion, dilution, and assimilative processes), were controlled.

The total waste flux between the two study sites were not significantly different but did correspond, in magnitude, to the slight differences attributable to the species of fish grown, their specific feeding rates, and their feed conversion ratios. For the Pacific salmon farm waste flux was estimated at 17.11 g/m^2/day while that for the Atlantic site was 18.35 g/m^2/day. These fluxes, which are equivalent to annual inputs of 6.45 – 6.70 kg/m^2/yr, are comparable or slightly lower than historical estimates that have been reported for such inputs. Those values have ranged from 12.6-24.2 kg/m^2/yr at a number of eastern Canadian farm sites (Hargrave, 1993), to 17.6–36.9 g/m^2/day in New Zealand (Morrisey et al., 2001), and <18.8 g/m^2/day for six farm sites studied by Brooks (2001) in coastal British Columbia. Nash (2001) recently claimed that a typical farm site, at current production levels, would theoretically release 25.7 g/m^2/day averaged over the entire production cycle.

The variability in organic waste flux estimates can be attributed to a number of factors. First, the organic material collected in sedimentation traps has been shown to degrade over time. Extrapolating from laboratory evaluations, Kelly and Nixon (1984) predicted that fresh deposits of organic material, collected in sediment traps, could degrade at a rate as high as 5-10%/day. Given this process, canister deployment duration becomes an important consideration in comparing waste flux estimates across studies. This methods constraint could also result in an underestimation for depositional rates.

Although minimal waste material was likely lost from the canisters designed for this study, an indeterminate proportion may have been liquefied as a result of a degradation process once the material had accumulated within the containment devices. In a recent study that focused specifically on salmon faecal material, Chen et al. (2003) estimated that nutrient leaching from faecal material occurs during the first 2-5 minutes following emersion in seawater and that there is no significant, additional leaching following this initial contact period. Furthermore, faecal settling rates documented by Chen et al. (2003) were in the range of 3.7–9.2 cm s^{-2}.

Using both of these estimates, faeces released to the seawater in this study would have theoretically leached its nutrient load within a range of 4.4 and 27.6 meters, or within an average settling distance of 16.04 meters. Given that all but the in-cage canisters were situated at or below the 15-meter depth plane in this study, it is likely that most of the soluble waste was released to the seawater as a result of this initial degradation process and prior to accumulation at the established sampling stations.

The differing organic waste flux estimates reported over the past decade also reflect the changes that have occurred in the salmon aquaculture industry as it has matured over this period. Technological innovation, for example, has been applied to feeding practices (camera and acoustic feed monitoring systems) to reduce loss through wastage. Feed formulations have been modified to accurately meet the dietary requirements of the growing fish, which has been directly responsible for reducing Feed Conversion Ratios (FCR's) and thus the proportion of organic loss to the environment as faeces.

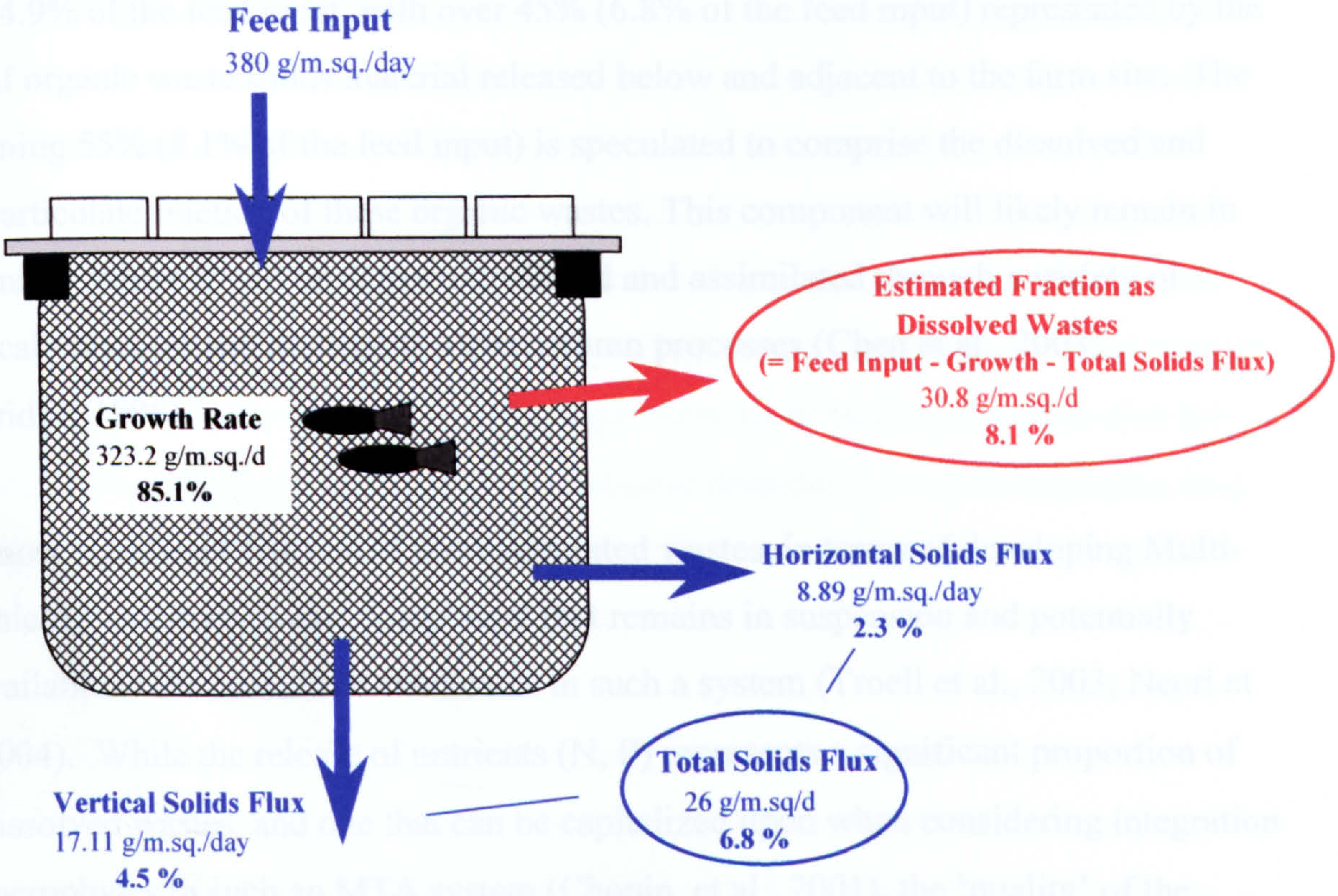
An understanding of the sedimentation and dispersion processes of farm wastes has been recognized as an important consideration in defining zone of influence, non-target species impacts, the environmental persistence of waste constituents, and hence the ecological 'foot-print' that is necessary in establishing appropriate farm siting criteria (Levings, 1994; Sutherland, 2001). These processes are also pivotal to understanding the potential interactions of species considered as co-culture candidates for a potential integrated aquaculture system.

The loss of organic waste material from the two study sites confirm previous observations (Sutherland, 2001; Gowan and Bradbury, 1987) that a significant proportion of these wastes (solids component) are lost vertically, through the bottom of the net-cages, with a significantly lesser fraction transported through the cages and along a horizontal pathway. The horizontal flux (corrected for background organic input) was estimated using the collected waste material at the mid-water station immediately outside (and downstream) of the cage system. At the study site with low tidal flow influences this flux was 8.98 g/m^2/day, or approximately 2.3% of the feed delivered to the net-cage directly adjacent to this sampling station.

A mass balance model of the total organic material cycled through the net-cage system can be developed using the farm operational information and waste canister data acquired during this study component. Feed represents the organic matter input to the model, the fish serve as an organic matter reservoir, and the waste is organic matter lost from the system as the difference between the former two. Figure 52 summarizes this relationship in a simplified diagram that illustrates feed input, the retention of organic material in the growing fish stock during growth, the fate of solid waste transport vertically and horizontally from the cage system, and the proportion of waste that remains unaccounted for in such a model.

Figure 52:

Mass balance diagram (Young Passage study site) examining input of feed and partitioning of this biomass from fish growth through waste material loss. Data are standardized as rates/fluxes (m^2 of surface inputs and discharge flux) and are based on measured operational variables (feeding rate, fish growth) and the waste flux data acquired from the canister component of the study (all shown in blue). An estimate of biomass loss through the dissolved (or suspended particulate) pathway is portrayed in red. The assumptions and basis of the inputs/fluxes are presented in the bottom table.



Data Used in Mass Balance Diagram:

Study Site	Young Passage
Feed Rate:	actual records - average over 10-month canister deployment period 342.4 kg/cage/day
Number of Fish:	average/cage over 10-month deployment period 490,295 fish at site n = 40,858 fish/cage
Growth Rate:	actual records - weekly over 10-month deployment period Fish Weight (g) = 49.851x + 691.53 r.sq. = 0.9489 Daily Growth Rate = 7.12 g
Total Daily Growth:	290.91 kg/cage/day = 323.2 g/m.sq/day assuming equal distribution of fish in cage
FCR:	1.18

The blue portions of the mass balance diagram represent components that were determined through direct measurement, and hence provide some certainty in the resulting estimations. The red component, comprised of the dissolved and fine suspended particulate fraction, was derived from the other two. The assumptions of this model are provided at the bottom of Figure 52, and suggest that this salmon production cycle was typical of current industry operations with an efficient feeding regime that resulted in minimal organic waste.

With a calculated Feed Conversion Ratio (FCR) of 1.18, the total loss of organic waste was 14.9% of the feed input, with over 45% (6.8% of the feed input) represented by the flux of organic waste solids material released below and adjacent to the farm site. The remaining 55% (8.1% of the feed input) is speculated to comprise the dissolved and fine particulate fraction of these organic wastes. This component will likely remain in suspension where it will be dispersed, diluted and assimilated through a variety of physical-chemical and biological water column processes (Chen et al., 2003; Beveridge, 1996).

The most significant fraction of farm-generated wastes, in terms of developing Multi-Trophic Aquaculture, is that component that remains in suspension and potentially bioavailable to the co-culture candidates in such a system (Troell et al., 2003; Neori et al., 2004). While the release of nutrients (N, P) represents a significant proportion of the dissolved wastes, and one that can be capitalized upon when considering integration of macrophytes in such an MTA system (Chopin, et al., 2001), the 'quality' of the growing water, in terms of other waste constituents, must also be considered as a potential risk to the co-culture species. Specifically, the proportional release of trace metals from feed wastes and the clearance of chemotherapeutic compounds following periodic treatments will both affect the near-field growing water quality proposed for a candidate co-culture species.

In this study the trace metal constituents of faecal wastes were found to provide a useful inferential indicator of spatial impacts of farm-derived organic material. Although individual trace metals can not be used, in isolation, to unequivocally differentiate source (Sutherland, 2001), the proportion (sample composition) of these metals did provide a clear spatial relationship that followed that of the total waste flux signal. Carbon, calcium, phosphorus, zinc, copper, cadmium, and strontium were correlated with farm waste discharge, and revealed significantly elevated levels at the base of the cage systems. Levels of these trace metal constituents declined with distance downstream of both study sites, with the dispersion pattern significantly influenced by the oceanographic and physiographic attributes of the sites.

Other research (Hansen et al., 1991) has argued that stable carbon isotope composition of feed could provide a unique signature of farm wastes, and have suggested that the isotope composition reflects the specific lipid, protein and carbohydrate sources used in manufacturing the feed and hence can be used to detect sediment impacts associated with farm waste inputs. However, in comparing the feed signature of stable carbon isotopes with that of collected wastes beneath salmon net-cages, Sutherland et al. (2001) found that an absence of the feed-specific carbon isotope signature was most likely attributable to fractionation that occurred during digestion, a lack of sufficient feed pellet waste to detect the signal, and/or a dilution of the signature by other sources of organic matter in the samples. Again, improvements in feeding practices over the past ten years, including use of camera or acoustic detection devices to minimize feed wastage, has eliminated feed as a major source of organic material loss from a farm site, which now comprises primarily fish faeces.

The trace metal signature in the organic wastes, measured as far as 60 meters downstream of the study farm sites, suggest that these waste constituents could be available to co-culture candidates considered for an MTA system. While the metal levels were reportedly very low, the risk of uptake by non-target species is not likely the direct effects at these concentrations but the potential for bioaccumulation or bio-concentration that could lead to taxon-specific toxicity or, in the case of a species used for aquaculture purposes, levels that exceed seafood safety criteria (Huang, 2003).

However, assessing the environmental flux of these elements is difficult given that fish can absorb and excrete trace metals directly from/to the surrounding water (Jobling, 2001), or from dietary sources (feed), at varying rates as required through the life cycle of the fish (Richardson, 1986; Lorentzen and Maage, 1999). Estimating the flux and potential environmental effects of this component of finfish aquaculture waste are further complicated by the tendency for zinc, and other trace metals constituents of feed, to bind to other feed components and to become unavailable to the fish and hence released with faecal waste material (Lall, 1991).

Zinc and copper were two trace metals that provided a clear environmental flux in the waste collection canisters deployed at both of the study sites. Both elements represent important dietary requirements for salmon growth and physiological function. While a minimum dietary requirement for copper has not been reported, Maage and Julshamn (1993) determined that zinc is required in the range of 37-67 mg/kg.

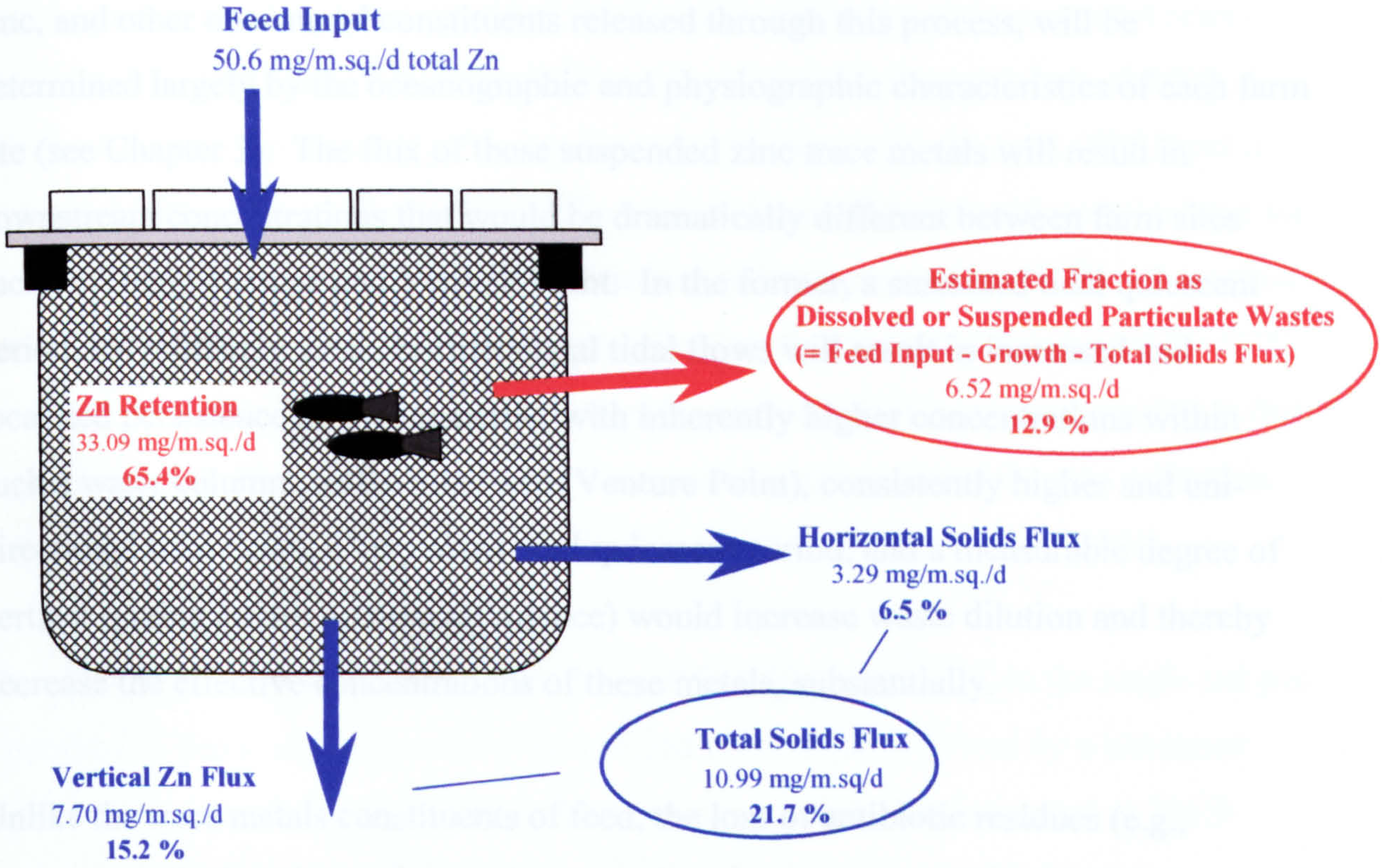
To further assess the potential for trace metal release to the environment, and their bioavailability (and risk) to an integrated aquaculture system, a mass balance model similar to that presented for the total organic waste was prepared for zinc. It was assumed that similar such processes would apply to other trace metal constituents, although environmental concentrations may vary depending upon the differential retention of the individual micronutrients.

Figure 53 presents the total zinc mass balance model for the Young Passage farm site. A concentration of 133 mg/kg of dietary zinc was used as an input value for feed used at this study site (derived from a sample analysis during the study). Retention of zinc in the fish was estimated using a value of whole-body zinc concentration reported by Lorentzen and Maage (1999) following application of a diet comprised of 138 ± 7 mg/kg zinc, comparable to the diet supplied in this study. Additional zinc retention was attributed to the unconsumed feed that is typically retained in farmed fish stomachs; a average of 3.5 g of feed was measured from a number of fish stomachs (n=10) and hence used to represent this component of the mass balance model.

Using these parameters, the input rate of zinc to the salmon net-cage system was estimated at 50.6 mg/m^2/d. Over 65% of this zinc was subsequently retained by the resident fish, with approximately 11% lost with the settleable wastes that were quantified through the canister component of this study. It was also estimated that 70% of the zinc retained in the organic waste was dispatched through the vertical loss to the seafloor while the remaining 30% was transported laterally from the cage system.

Figure 53:

Mass balance diagram (Young Passage study site) examining input of zinc and partitioning of this element from retention in the fish during the growth cycle and through loss to the environment. Data are standardized as rates/fluxes (m^2 of surface inputs and discharge flux) and are based on measured operational variables (feeding rate, fish growth) and the waste flux data acquired from the canister component of the study (all shown in blue). An estimate of zinc retention, and zinc loss through the dissolved (or suspended particulate) pathway, are portrayed in red. The assumptions and basis of the inputs/fluxes are presented in the bottom table.



Data Used in Mass Balance Diagram:

Study Site	Young Passage
Feed Rate:	actual records - average over 10-month canister deployment period 342.4 kg/cage/day
Dietary Zn:	133 mg/kg feed (actual delivered)
Growth Rate:	Daily Growth Rate = 7.12 g/fish (see Figure 52) = 290,909 g/cage/d Zn retained (whole animal): 41.7 mg/kg (from Lorentzen and Maage, 1999) Mean feed retention in gut = 3.5g (observation/measurement) = 430.5 ug Zn/fish
Total Zn Consumption:	29740.70 mg total Zn retained in fish = 33.05 mg/m.sq./d, assuming fish are evenly distributed across net pen

The zinc fraction that was unaccounted for in this analysis represented 12.9% of the zinc delivered to the cage system on a daily basis. This fraction is speculated to represent the zinc that is bound to a fine particulate component of the discharged waste, and hence that material which would remain in suspension and be available for horizontal transport downstream of the farm. This fraction would also include the dissolved zinc component that may be excreted through urine, skin or gill tissues.

It is this waterborne component that may provide the greatest risk to adjacent, non-target species and/or to species that are introduced adjacent to a finfish system as part of an integrated aquaculture system. The subsequent environmental concentrations of zinc, and other trace metal constituents released through this process, will be determined largely by the oceanographic and physiographic characteristics of each farm site (see Chapter 3). The flux of these suspended zinc trace metals will result in downstream concentrations that would be dramatically different between farm sites such as Young Passage and Venture Point. In the former, a sustained tidal quiescent period and relatively weak, bi-directional tidal flows will result in increased and localized persistence of such material, with inherently higher concentrations within such a water column. In the latter site (Venture Point), consistently higher and uni-directional tidal flows, a very short tidal quiescent period, and a measurable degree of vertical mixing (water column turbulence) would increase waste dilution and thereby decrease the effective concentrations of these metals, substantially.

Unlike the trace metals constituents of feed, the loss of antibiotic residues (e.g., oxytetratcycline) to the marine environment is a short-term event that is associated with a specific treatment period. The application of oxytetratcyline (OTC) was required on four occasion over the salmon production cycle at Young Passage. Although the focus of this study was on the potential bioaccumulation of these residues on adjacent shellfish resources, the deployment of canisters to establish waste material flux at these study sites did provide an opportunity to examine the flux of OTC residues within the waste materials of this farm site.

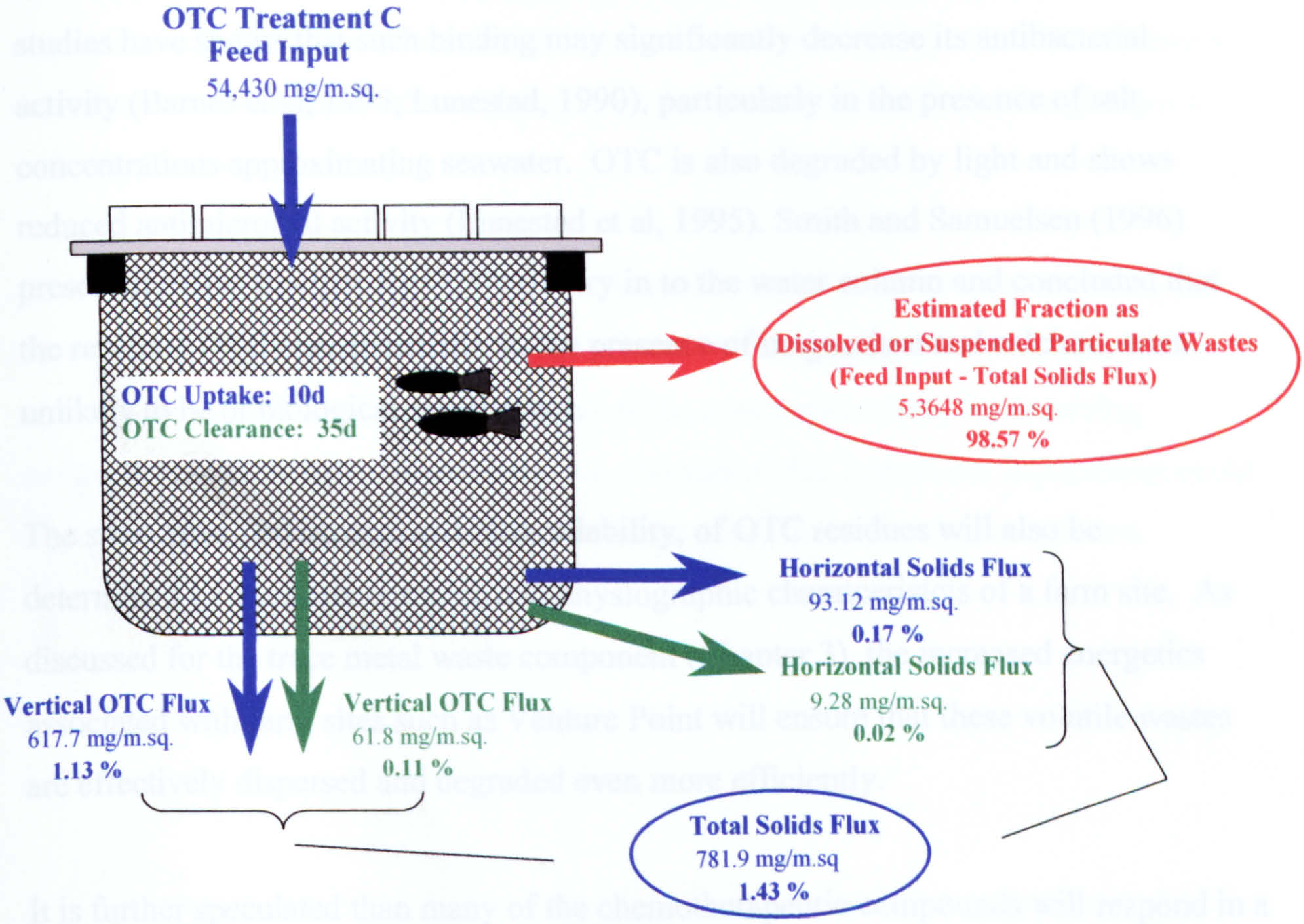
The results of this assessment found that OTC residues associated with settleable wastes were dispersed downstream of the farm site to a distance no greater than 125 meters for bottom canisters and 75 meters within the water column. Given the multiple treatments for this site, collected waste material revealed that OTC clearance from the farm occurred with 30-45 days at which time no additional flux of OTC could be measured from the system. Although persistence of this material on the seafloor would occur well beyond this time frame (Coyne et al., 2001; Jacobsen and Berglind, 1988; Samuelsen et al., 1992;. Jacobsen and Berglind; 1988; Samuelsen, 1989; Pouliquen et al, 1992), the duration of release from the net-cage and into the surrounding environment, as solids and/or as a dissolved/suspended component, are important consideration for assessing the potential interactions with non-target organisms within the influences of the water column.

In an approach similar to that provided for the total waste flux and trace metal release, the analysis of OTC fate (flux as mg/m^2) was developed for the total dose of OTC delivered during Treatment C at Young Passage. In Figure 54, the total OTC input is compared with corrected waste flux, and a mass balance of these residues calculated for this single medicated treatment. The corrections to the measured waste flux comprised a standardization accounting for sediment collection that preceded the start of the OTC treatment, as well as an estimated correction for probable OTC residue degradation that would have occurred within the sediment traps during the prolonged waste collection period (26 days). The latter used half-life data reported in Coyne et al. (2001).

The entire treatment resulted in the delivery of 54,430 mg/m^2 OTC in the single net pen examined in this study. Released over a period of 10 days, followed by a clearance period of 35 days, the loss of OTC residues as waste solids was estimated at 781.9 mg/m$_2$ and was equal to 1.43% of the total compound delivered to the system as medicated feed. In the study completed by Coyne et al. (2001) the residues detected in bottom sediments, invertebrates and the salmon could not account for more than 1.4% of the total input of OTC. The authors concluded that the antibiotic was very rapidly dispersed and assimilated within the environment, and that it could not pose an environmental or human health risk as a result.

Figure 54:

Mass balance diagram (Young Passage study site) examining input of OTC during Treatment C and fate of this antibiotic from delivery in feed, to the fish and through to loss to the environment. Data are standardized as fluxes (m^2 of surface inputs and discharge flux) and are based on measured operational variables (treated feeding rate, duration) and the waste flux data acquired from the canister component of the study. Estimated flux during treatment shown in blue, clearance period in green, and OTC loss through the dissolved (or suspended particulate) pathway in red. The assumptions and basis of the inputs/fluxes are presented in the bottom table.



Data & Assumptions Used in Mass Balance Diagram:

Study Site	Young Passage
Medicated Feed Rate:	35,500 kg medicated feed
OTC Degradation:	canister deployment was 26d; loss of residues was compensated by using a hindcast to predict actual flux; 5d half-life used as difference between sediment and mussel residue half-lives reported by Coyne et al., 2001
OTC Treatment:	4.89 kg active ingredient/cage/day (1.67% OTC)
Treatment Duration:	10 days
Clearance Period:	35 days; calculated as canister levels < 1.0 ug/g OTC

The treatment examined in this study suggests that over 98% of the OTC remains unaccounted for in a mass balance assessment. It is speculated that this material is largely eliminated from the farm system as a dissolved component, or adsorbed to fine particulate matter that remains suspended during the subsequent degradation process. Remaining in the upper water column, these materials may become bioavailable to non-target organisms located within the near-field region around the farm structures, but persistence (longevity) in the water column is limited.

Once in solution OTC has been shown to bind heavily (95%) to divalent cations such as with magnesium (Hektoen et al, 1995; Lunestad and Goksyor, 1990). Experimental studies have shown that such binding may significantly decrease its antibacterial activity (Barnes et al, 1995; Lunestad, 1990), particularly in the presence of salt concentrations approximating seawater. OTC is also degraded by light and shows reduced antimicrobial activity (Lunestad et al, 1995). Smith and Samuelsen (1996) presented model kinetics for OTC re-entry in to the water column and concluded that the resultant OTC concentrations, in the presence of magnesium and calcium, were unlikely to be of biological significance.

The short-term persistence, and bioavailability, of OTC residues will also be determined by the oceanographic and physiographic charatceristcis of a farm site. As discussed for the trace metal waste component (Chapter 3), the increased energetics associated with farm sites such as Venture Point will ensure that these volatile wastes are effectively dispersed and degraded even more efficiently.

It is further speculated than many of the chemotherapeutic compounds will respond in a manner similar to that documented for OTC or for other agents such as furazolidone (Samuelson et al., 1991). Most of these compounds are applied in very small quantities or volumes (active ingredient), are sensitive to environmental factors such as light, and represent short temporal fluxes that facilitate efficient environmental assimilation of the waste constituents.

The application of emamectin benzoate (Slice ^{TM}) at the Venture Point farm site, and the subsequent opportunistic efforts to assess environmental loading of this compound using the sediment trap samples, may support this premise. Although only a single set of sediment trap data were analysed for emamectin benzoate residues, no evidence of such residues were found in any of the downstream samples for this farm site.

Upon further investigation of these analytical data it was assumed that the level of detection for this residue was set too high (<0.1 ug/g), and that the emamectin benzoate signal was in fact missed in these waste samples. Nevertheless, the environmental fate of these residues that are reportedly released at ppb levels (Schering-Plough, 2002; Willis and Black, 2003) would be equivalent, or likely much less, than that presented for the OTC scenario. Emamectin benozoate, the active ingredient of Slice ™, was delivered to the study system at a dosage of 5.72 g/cage, or 6.36 mg/m^2 – approximately 8,500 times less than the active ingredient input for the OTC treatment. At this treatment dosage, delivered over a 10-day period, the individual cage input is estimated at 636 ug/m^2/day.

Given that this short-term chemotherapeutic treatment occurred at Venture Point, the rigorous tidal mixing would have been sufficient to thoroughly dilute this material and enhance the degradation and assimilation of the compound within the receiving environment. It is highly improbable that residues of this compound is persistent could be easily detected in the receiving environment given these physical conditions.

CHAPTER 5

Contaminant Effects on Shellfish

5.1 Introduction

The composition of finfish aquaculture feed includes a mixture rich in proteins, carbohydrates, lipids, as well as a dietary balance of essential vitamins and minerals (Lall, 1991). While there is a wealth of literature describing the environmental impacts of organic wastes originating from the digestion and excretion of wastes from these feeds (see Chapter 4), there is comparatively little research exploring the potential environmental fate and effects of the minor feed constituents that represent a continual input (e.g., micro-nutrients) or periodic supplements to the feed (e.g., antibiotics, chemotherapeutics).

The release of feed additives will be conveyed to the receiving environment along two distribution pathways. Discharged within the solid waste component, as fish faeces, the organic wastes containing these constituents will be dispersed with the tidal energy and flow characteristics specific to each farm site, and will accumulate over the seafloor in a spatial pattern defined by these physical processes (Chromey, 2001). The accumulation of these wastes will result in a direct and indirect impact on benthic communities, with the former including effects such as smothering, decreased bottom oxygen, a shift in geochemical assimilative processes (Brooks, 2001; Hargrave, 1997) that leads to interstitial toxicity (e.g., elevated sulfides), etc. Indirect impacts may result in areas of lower organic material accumulations, with subtle changes to benthos structure realized through ecological processes such as competitive exclusion and/or opportunism that are typical of gradients of organic enrichment (Pearson and Rosenberg, 1978; Weston, 1990).

The second pathway includes the lateral dispersion and dilution of these constituents to the water column as fine particulates that remain in suspension (TSS), or as the dissolved (soluble) fraction that is released from the farm site through urinary excretion or as a result of the breakdown of faeces (Chen et al. 2003). The persistence of these constituents will be determined, again, by the physical oceanographic conditions unique to each farm site. The tidal energy and site-specific physiographic characteristics will influence the dilution and dispersion processes, and hence the effective concentration of such contaminants in the water column. Chapter 3 discusses these processes, and their importance to the fate of the settleable and suspended solids, as well as the dissolved (soluble), components of waste discharged from finfish aquaculture facilities.

Exposure and/or potential accumulation of these waste constituents in non-target organisms represent an important, albeit secondary category of environmental effects to that of the solid waste discharges and their associated seafloor impacts. Given the large proportion of organic waste, and its trace metal and antibiotic constituents, that are estimated to remain suspended (or dissolved) within the water column (13% for metals, 98% antibiotic residues: see Chapter 4, Section 4.5), the fate and subsequent environmental effects/risk of these contaminants should be fully understood.

Trace Metals (Micro-Nutrients)

The release of trace metals with faecal wastes has been shown to accumulate on the seafloor and concentrate to levels that could result in a negative impact on localized infaunal and epifaunal assemblages (Brooks, 2001). Despite an understanding of the dynamics in waste flux from fish farm facilities (Chromey, 2002), and more recently the composition (Sutherland et al., 2001; Chen et al., 1999) and the spatial extent of these cumulative impacts (Weston, 1990), most of the research has examined the overall community structural change which has occurred in these faunal communities (Brooks, 2001).

Comparatively little research has been applied specifically to the response of non-target species to the trace metal constituents of these organic wastes, whether in the benthic environment or the water column. Chou et al. (2002) measured the lobster tissue (digestive gland) concentrations of selected trace metals beneath a salmon farm facility and at a number of sites well removed from the influences of such operations. The concentration of copper was an order of magnitude higher at the farm site (133 ug/g versus 12.6 ug/g at a reference site), although there was no statistical difference among samples analyzed for iron, manganese, or zinc.

Chou et al. (2003) completed a study that used metals in green urchins (*Strongylocentrottus droebachiensis*) as indicators for the near-field effects of chemical wastes originating from salmon farm operations in eastern Canada. Subtle change in growth performance in these urchins, and elevated trace metal concentrations (copper, zinc, iron, magnesium, cadmium) in the intestine tissue indicated a quantifiable downstream impact zone extending to 75 meters of the arm site.

Although the study of trace metal transfer from aquaculture facilities to non-target organisms are limited, the effects of trace metals (from naturally elevated waters or other pollution sources) on aquaculture species has been presented. Liao and Lin (2001) studied the toxicokinetics and acute toxicity of zinc in abalone, and found that these taxa can uptake and depurate zinc very quickly. Lin et al. (1993) revealed that filtration rates in oysters (*Crassostrea gigas*) could be inhibited by exposure to elevated concentrations of cadmium or zinc.

Wong et al. (2001) found that poor water quality in Hong Kong resulted in elevated trace metals levels in a number of commercially cultured fish species, with particular concern over levels of both copper (which accumulated in liver tissues) and zinc which concentrated in gonad tissue. Han et al. (1994) documented coastal levels of trace metals in Taiwan, relating elevated levels of copper to seafood safety concerns over shellfish produced in the region. Levels in excess of 4,400 ug/g resulted in widespread closures in the shellfish industry and a significant financial loss; the reported levels were sufficient to cause a green discoloration in oyster tissues.

Many bivalves species have been used as indicators of coastal pollution due to their ability to sequester and store trace metals. Mussels, oysters and to a lesser extent clams, have all been employed as bioindicators in such programs with considerable success (Goldberg, 1986; Laurenstein et al., 1990; Castro, et al., 1996). The use of shellfish in the present study allows a similar approach to evaluating potential trace metals effects in the water column downstream of a potential impacting source.

Chemotherapeutic Compounds

Chemotherapeutic compounds represent another group of potential contaminant constituents originating from feed used at finfish aquaculture farms. As with antibiotics, many of the chemotherapeutic compounds are milled directly into the feed, at a prescribed dosage, and delivered to the fish as a medicated treatment when required of a production cycle. A common example of this class of compounds is the pesticide emamectin benzoate, which is offered as the commercial sea lice treatment drug referred to as Slice ™.

Salmon readily absorb emamectin benzoate from the medicated feed, and release to the environment typically occurs through faecal material or via wasted (uneaten) feed. Excretion of the metabolites occurs for an extended, post-treatment period, and given the insoluble nature of the compound and its affinity to particulate matter, most of this waste material settles to the seafloor where it becomes a localized risk to the epifaunal and infaunal benthos (SAMS, 2003).

SEPA (1999) conducted a thorough environmental risk assessment for emamectin benzoate and also concluded that the compound is likely to remain tightly bound in sediments due to its low seawater solubility. The study further summarized the break-down and degradation characteristics of the compound in sediments, suggesting that under aerobic conditions the half-life was 193.5 days and approximately 427 days under anaerobic conditions. However, in the presence of light and contact with microbially active sediments the half-life was as little as 5 days.

Little scientific research has addressed the issue of non-target species effects for this group of chemotherapeutic compounds. However, it has been shown that crustaceans, and primarily the early developmental stages, are the most sensitive to these pesticides (Burridge et al., 2000). A recent study, focused on the non-target species effects of Slice (emamectin benzoate), reported that nominal exposure caused female lobster (*Homarus americanus*) to molt prematurely (Waddy et al., 2002).

Another study (Willis and Black, 2003) suggests that there is a toxic response in non-target copepod species, but this response was observed at emamectin benzoate exposure levels considerably higher than the proposed Environmental Quality Standard (EQS) proposed by SEPA for this compound. The study concluded that an adverse affect on copepod populations would be unlikely.

Antibiotics

There is a wealth of literature that describes the pharmacokinetics of oxytetracycline (OTC) used in finfish aquaculture. These studies have spanned a range of species, including charr (Haug and Hals, 2000), salmon (Abedini etal., 1998; Namdari etal., 1999), sea bass (Rigos et al., 2002), and sea bream (Rigos et al., 2003). These initiatives have examined drug partitioning across fish tissues, the efficacy of treatment dosage, and the effects of ambient conditions on drug performance. Similar such research has been conducted for invertebrate culture species, including the shrimp *Panaeus stylirostris* (Mohney et al., 1997).

The environmental concerns related to antibiotic usage in open systems typically include the potential for developing antimicrobial resistance. Numerous freshwater finfish systems have been studied for such effects (Guardabassi et al. 2000; Dalsgaard and Madsen, 2000; Schmidt et al., 2000; Bruun et al., 2000) as well as at marine finfish facilities (Miranda and Zemelman, 2002; Schmidt et al., 2001; Chelossi et al., 2003). The bacterial loads and resistance to OTC applications in shrimp ponds have also been evaluated (Tendenci and dela Pena, 2002; Hameed et al., 2003).

The loss of OTC residues from marine net-cage operations also represents an important contaminant, in a traditional discharge sense, as it becomes available to a range of non-target species living in the water column or bottom sediments once released (through faeces, urine) from a farm facility. The present study has suggested that over 98% of the residues become available in a dissolved or suspended form (Chapter 4), and thus available to non-target species located downstream of such a facility. Of particular concern are those non-target species that are consumed by humans (e.g., fish, clams, crabs, prawns, etc.). The Canadian Food Inspection Agency (CFIA) has placed a "safe" level for OTC in seafood at 0.1 ppm. In the United States this level is set at 2.0 ppm.

OTC residues have been detected in wild fish up to 13 days post medication (Bjorklund et al, 1990) and in wild and cultured shellfish in the vicinity of fish farms after medication (Coyne et al., 1997; Hansen et al, 1992; Lunestad, 1992, Sameulsen 1992 and 1993; and Ervik et al, 1994; Pouliquen et al, 1993). Capone et al. (1996) and Weston et al. (1994) found trace OTC residues (0.1ug/g) in oysters (*Crassostrea gigas*) and Dungeness crab (*Cancer magister*) but found red rock crab (*Cancer productus*) with levels of up to 3.8 ug/g 12 days after an OTC treatment at one site. Tibbs et al. (1989) and Peterson et al. (1993) were unable to show contamination of bivalves (*Crassotrea gigas* and *Mytilus edulis*) placed in seawater containing OTC or tetracycline, but they used antibiotic activity as the detection method. Pouliquen et al. (1992) showed that other methods were less sensitive and less specific than HPLC.

Campbell et al., 2001 examined the residence time of oxytetracycline in sea urchins (Psammechimus miliaris) as a potential factor limiting development of this as a co-culture species in an integrated salmon-urchin polyculture system. Their work, conducted under laboratory conditions, suggested that gonad tissue accumulation of OTC was highly variable and that the length of the elimination phase for these residues was considerable (half-life = 24.6 days at T=11-13° C).

Coyne et al. (1997) evaluated the persistence of OTC in blue mussels (*Mytilus edulis*) during and after therapeutic treatment of fish at a farm. They found no detectable levels of OTC in mussels 20 meters from the cage block, at a depth of 1 meter. The only groups to show significant levels of OTC were those immediately under a treated cage; however, at the end of treatment these levels declined exponentially with a half life of approximately 2 days. They concluded that residues present in filter feeding bivalves that occur as a consequence of the therapeutic use of OTC in marine fish farms are unlikely to present a significant human health hazard. The study did not, however, examine the dispersion of OTC at other points in the water column (e.g., mid net-pen, or beneath the net-pens) and the design may thus have missed the opportunity of documenting the pathway for OTC dispersion from a farm site.

In British Columbia, Black et al. (1991) used laboratory experiments to determine the potential for transfer of therapeutics from fish farms to oysters. They found that uptake by *Crassostrea gigas* occurred with 24 hours of exposure to OTC and that these levels were reduced from 18 ppm to non-detectable levels within 13 days. Le Bris et al. (1995) showed that *C. gigas* still contained significant levels of OTC 14 days post medication under experimental conditions. Experimental work with *Crassotrea gigas* and *Mytilus edulis* (Pouliquen et al., 1996) did show species differences. OTC concentrations in mussel tissues were always higher than in oyster tissues, though half lives and mean residence times of OTC in both demonstrated linear elimination kinetics.

The effect of physical oceanographic conditions on the downstream bioavailability of these compounds has not been documented. Similarly, an estimation of *in situ* uptake and non-target species clearance rates for antibiotic residues would provide a useful basis for examining integrated aquaculture potential from a seafood risk perspective. Consideration of these processes in the development of MTA, and the management of these inherent environmental risks, are critical to the success of this aquaculture innovation in temperate maritime regions.

5.2 Objectives

The loss of organic waste material from salmon net-cage systems has been shown to contain measurable levels of trace metals, delivered with feed and released through fish faeces and urine, with an estimated 12.9% of zinc (for example) that remained unaccounted for in a mass balance evaluation of the fate of such constituents (Chapter 4). Similarly, the treatment of fish with chemotherapeutic compounds such as oxytetracycline and emamectin benzoate, although representing a short-term input to the finfish culture system, resulted in a potential waterborne fraction that represented over 98% of the initial active ingredient contained within the feed.

Given the potential of these waterborne contaminants to negatively impact non-target species, and specifically those that might be considered in the development of an integrated aquaculture system, the objectives of this study component included:

- Evaluation of the tissue burden levels sustained in shellfish (scallop and oyster) grown adjacent to a marine finfish aquaculture facility over an entire production cycle;
- Assessment of the spatial and temporal limits of any tissue effects resulting from these contaminant loads; and
- Determination of the extent of tissue partitioning of these contaminants and their potential for impacting tissue quality and 'safety' from a human consumption perspective.

Results of these analyses were subsequently discussed in terms of the technical feasibility of developing integrated finfish-shellfish aquaculture, and the possible management strategies that could mitigate any of the tissue impacts that would represent a human health risk.

5.3 Materials and Methods

The evaluation the bioaccumulation of contaminant constituents in shellfish tissue was completed using a combination of sample collection, chemical analyses, and data evaluation procedures. The following sections describe these approaches for this study component.

5.3.1 Shellfish Species

The shellfish used for this assessment included *Crassostrea gigas*, the Pacific oyster, and *Patinopecten yessoensis*, the Japanese scallop. Both species are commercially important suspended (off-bottom) aquaculture species, ensuring that results of this research could be applicable from a commercialization perspective (i.e., development of integrated finfish and suspended shellfish aquaculture employing these species).

A further description of the rationale for selecting these species, as well as the infrastructure

5.3.2 Field Sampling Program

The shellfish were deployed from a commercial longline system established downstream at each of the two study sites (see Chapter 2). At each of the ten sampling stations positioned along this line scallops were contained on two droplines of Pearl Nets. Each vertical dropline comprised a set of ten nets, which were evenly spaced and deployed vertically across a depth range of 6 through 15 meters.

The upstream scallop dropline (closest to the farm system) was initially stocked with larger individuals (5-8 cm). These individuals were routinely sampled (sacrificed) and analysed for contaminant loads during the first half of the study. Once the first line was depleted of animals (due to removal for analytical purposes), the scallops on the second line (used for the culture performance component of the study – Chapter 6) were considered of sufficient size to continue the sacrificial sampling.

A single vertical dropline for oysters was also deployed at each station. A stack of four oyster trays was positioned in the upper layer of the water column, spaced evenly between depths of 1.0 and 6.0 meters. These animals were of sufficient size (4.11 ± 0.32 cm) that animals could be drawn randomly from any of the trays for the routine sampling component of the study.

Shellfish were sampled at each of the two study sites concurrently throughout the study, at an interval of approximately 45 days (which varied occasionally due to weather or anticipated conflict with farm operations). This routine sampling program consisted of extraction of ten scallops and five oysters from each downstream sampling station with on-site preparation for submission to the analytical laboratory. Individual animals were acquired randomly from the nets/trays at each of the sampling stations to eliminate possible depth-related effects, and then submitted as a composite for chemical analysis.

Upon retrieval, scallops were dissected to provide three tissue types for sample analyses, i.e., adductor muscle, roe/gonad, and other viscera. Oysters were submitted whole. Tissue samples was placed in sterilized glassware, packed in coolers maintained at approximately 4° C, and transported to a third-party analytical laboratory for the required chemical analyses. ALS Laboratories of Vancouver, British Columbia, were contracted for this component of the study given their accreditation with Canadian regulatory agencies (including Canadian Food Inspection Agency - CFIA). It was anticipated that the involvement of such a facility ensured that results from these chemical analyses would be accepted by regulatory agencies. In particular, these results could have some influence on decisions regarding required changes to regulation that would permit integrated finfish-shellfish aquaculture in the future.

The chemical constituent analyses of the shellfish tissue samples included routine analysis (45-day sampling interval) for a suite of trace metals (25 metals; see Table 1, Chapter 2), incidental post-treatment evaluation of antibiotic residues (oxytetracycline and tetracycline) and pesticide treatment for sea lice (emamectin benzoate). The analytical procedures employed for these analyses are summarized as follows.

- Trace Metals

These analyses were carried out using procedures adapted from "Recommended Guidelines for Measuring Metals in Puget Sound Marine Water, Sediment , and Tissue Samples" prepared for the United States Environmental Protection Agency and the Puget Sound Water Quality Authority, 1995. Tissue samples were homogenized either mechanically or manually prior to a heated digestion with nitric acid and hydrogen peroxide. The digest was subjected to analysis by inductively coupled plasma - mass spectrometry, inductively coupled plasma - optical emission spectrometry, or atomic absorption spectroscopy.

- Hydrocarbons

The analysis for volatile fuels, such as gasoline, was accomplished using a procedure adapted from the analysis of sediments and soils for volatile organic compounds (VOCs). The homogenized tissue samples were first extracted with methanol and a portion of each extract then analyzed using a headspace sample introduction technique coupled with capillary gas chromatography and flame ionization and/or photoionization detection (GC/FID/PID).

The analysis for the less volatile fuels, such as kerosene and diesel fuels, was carried out using a procedure adapted from the analysis of sediments and soils for extractable hydrocarbons. The homogenized tissue samples were extracted with dichloromethane, concentrated at atmospheric pressure, subjected to florisil adsorption chromatography and then analyzed using capillary gas chromatography and flame ionization detection (GC/FID).

- Oxytetracycline (OTC) & Tetracycline Residues

The specific details of this procedure were developed, and remain proprietary, to ALS Analytical Laboratories. In general, these analyses were carried out using procedures adapted from "Official Methods of Analysis of AOAC International", 16th Edition, AOAC Official Method 995.09, "Chlortetracycline, Oxytetracycline, and Tetracycline in Edible Animal Tissues". Tissue samples were extracted with a pH 4 buffer and the crude extract then cleaned up on C_{18} solid-phase extraction column prior to analysis by liquid chromatography using a C_{18} column and ultraviolet detection.

- Ememectin benzoate

Emamectin benozoate analyses were conducted using procedures adapted from Yoshiti (2000) and from Riet et al. (2000) that describe an analytical approach for quantifying the chemical constituents of the avermectin family of pesticides. In general, these analyses comprised an extraction stage (using acetonitrile), clean-up on a C_{18} solid-phase extraction column, and analysis using a high performance liquid chromatography (HPLC) with fluorescence detection. Specifics of this analytical approach, including detection wavelength, were requested to remain proprietary by the analytical laboratory.

5.3.3 Data Analysis

Data acquired from this component of the research program were compiled in a working spreadsheet, for each study site, and subsequently extracted and summarized in a variety of tables and figures that illustrated tissue burden levels, by shellfish species, with respect to distance from the respective net-cage facilities. Data were also standardized to eliminate reference condition levels so that farm-related effects could be readily identified from these data.

The chemical constituents analyses conducted on the individual tissues were examined to assess potential tissue partitioning of chemical contaminants, and discussed in terms of product safety and culture limitations. Principal Components Analysis (PCA) was used to differentiate chemical composition among the tissue groups for each study site, and partitioning ratios developed to indicate trace metal constituents and specific scallop tissue(s) that could be at risk in a co-culture (MTA) development.

Temporal and spatial changes in tissue burden levels of antibiotic residues were assessed concurrently by submitting these data to a 3-dimensional contouring program (Surfer 7.0). The resulting plots were used to determine zone of influence for chemotherapeutic treatments, duration and magnitude of shellfish effects, and the potential impact on the development of finfish-shellfish MTA.

5.4 Results

The potential impact of waterborne contaminant transport from finfish farm facilities to non-target species was evaluated using two species of bivalve mollusks (shellfish), *Patinopecten yessoensis* (scallop), and *Craasostrea gigas* (oyster). The spatial and temporal impact of continuous discharged waste constituents (e.g., trace metals, or micronutrients of feed) on tissue levels in these shellfish was examined at a high energy farm site (Venture Point) as well as a site characterized by comparatively low tidal flows (Young Passage). Periodic inputs of chemotherapeutic compounds were used to assess the bioavailability and persistence of these waterborne materials to shellfish resources.

The accumulation and persistence of trace metals in shellfish tissues were assessed over a 17-month period (April/2002 through September/2003) at the Young Passage farm site and over a 1-year period starting in June/2002 at Venture Point. Oysters were deployed for a relatively short period at each site, spanning an 8-month interval starting in December/2002.

5.4.1 Trace Metals in Oyster Tissues

Table 16 summarizes the trace metal results acquired for the oysters sampled at the Young Passage (upper table) and Venture Point (lower table) study sites. These data represent sample means (n=3) for the 12 trace metal constituents that reported levels above their respective levels of analytical detection. The blue values provide tissue levels of oyster reference samples, while the results at the bottom of Table 16 summarize historical data collected by the Canadian Food Inspection Agency (CFIA) of samples acquired from aquaculture and wild harvest areas (oyster results from this source portrayed in blue).

Statistical tests (t-tests) for each trace metal constituent, comparing study site values with reference samples, indicated that there was no significant difference ($P<0.05$) for any of these parameters. Despite an indication of some clear spatial trends in specific trace metals (e.g., declining Cu and Al levels downstream of the cage system at the Young Passage site) none of these values were significantly higher than those in the reference material or than values reported historically by the CFIA.

Many of the trace metal levels found in oyster tissue (whole animal) were consistently higher at the Venture Point study site as compared with those samples acquired at the Young Passage farm site (Table 16). While copper revealed tissue concentrations that were 2-3 ug/g higher at Venture Point than at Young Passage, tissue levels of aluminum were almost double. Cadmium levels were slightly greater, averaging 0.5 ug/g higher at Venture Point, while strontium was 1-2 ug/g higher in these tissues. However, these levels were all similar to those found in the reference material and in the historical CFIA database.

5.4.2 Trace Metals in Scallop Tissues

Figure 55 illustrates the differential accumulation (partitioning) of 13 trace metal constituents in scallop tissues. The three analytical groups comprised adductor muscle (meats), roe (gonad), and the remaining visceral components of the animal (e.g., mantle, gut, gill). Samples across time were pooled (averaged) and used to illustrate spatial differences in each of these tissue components. The figure presents the results of a centered (arithmetic mean) Principal Components Analysis (PCA) that was used to assess these metals data concurrently.

Table 16:

Mean trace metal (ug/g) tissue levels (n=3) for *Crassostrea gigas* (whole) grown at Young Passage and Venture Point study sites. Reference values in blue, and historical shellfish data acquired for production and wild harvest areas of B.C. provided in bottom table (CFIA, 1999-2001; n=18).

Mean	Distance	Al	As	Cd	Ca	Co	Cu	Pb	Mg	Mn	Se	Sr	Zn
Young Passage (*Whole Oyster*)	**-15**	10.33	1.45	2.23	314.33	0.030	12.18	0.03	538.00	2.13	0.53	3.45	97.17
	0	8.67	1.51	2.31	413.00	0.023	12.50	0.03	540.67	2.81	0.57	3.67	111.57
	10	9.67	1.65	2.07	603.00	0.027	11.49	0.03	536.00	1.91	0.50	4.12	107.27
	20	8.67	1.59	2.09	320.33	0.027	11.01	0.03	533.33	2.29	0.50	3.45	117.67
	30	9.67	1.55	2.22	285.00	0.027	11.91	0.04	513.33	1.99	0.50	3.31	153.00
	50	9.00	1.58	2.36	547.00	0.027	11.68	0.04	528.33	1.94	0.53	3.90	152.33
	75	9.33	1.51	2.22	456.67	0.027	9.60	0.03	519.33	1.94	0.50	3.66	116.80
	100	8.67	1.53	2.20	378.33	0.023	8.89	0.03	534.00	2.08	0.50	3.66	92.97
	125	7.33	1.52	2.16	371.67	0.027	8.86	0.03	547.00	1.78	0.47	3.59	126.00
	175	8.67	1.52	2.23	287.00	0.027	9.60	0.03	526.00	2.09	0.50	3.15	125.33
	225	6.50	1.44	2.22	287.00	0.025	9.08	0.03	530.00	3.06	0.60	3.38	103.50

Mean	Distance	Al	As	Cd	Ca	Co	Cu	Pb	Mg	Mn	Se	Sr	Zn
Venture Point (*Whole Oyster*)	**-15**	17.63	1.50	2.16	475.33	0.028	13.65	0.038	586.67	2.46	0.48	4.45	111.13
	0	16.07	1.78	2.78	378.67	0.032	14.83	0.039	576.67	2.68	0.48	4.16	147.17
	10	13.23	1.47	2.48	416.33	0.028	14.61	0.035	635.00	2.06	0.50	4.62	132.33
	20	13.83	1.57	2.60	319.00	0.026	12.29	0.036	601.33	1.85	0.49	3.83	136.67
	30	18.27	1.58	2.68	435.67	0.032	13.47	0.041	619.67	2.63	0.51	4.36	146.33
	50	18.13	1.75	2.94	707.33	0.035	14.50	0.042	600.67	2.91	0.56	5.21	160.67
	75	19.93	1.57	2.68	416.00	0.032	13.31	0.039	631.33	2.75	0.53	4.53	139.53
	100	19.13	1.62	2.86	647.33	0.034	14.28	0.038	604.00	2.76	0.52	5.06	148.17
	125	18.30	1.58	2.82	644.00	0.031	13.44	0.041	623.00	3.13	0.55	5.02	140.00
	175	16.53	1.43	2.66	895.00	0.029	11.17	0.039	655.67	2.79	0.46	5.98	127.67
	225	18.73	1.60	2.87	409.67	0.033	13.44	0.043	599.00	2.45	0.62	4.48	163.67
Reference	Mean:	**12.00**	**1.51**	**2.43**	**489.89**	**0.029**	**10.46**	**0.036**	**549.22**	**2.53**	**0.54**	**4.00**	**127.67**
	Stdev	**7.91**	**0.28**	**0.36**	**388.63**	**0.011**	**4.19**	**0.012**	**59.55**	**0.72**	**0.14**	**1.34**	**30.63**
	n	**9**	**9**	**9**	**9**	**9**	**9**	**9**	**9**	**9**	**9**	**9**	**9**
	95 CL	**5.16**	**0.19**	**0.24**	**253.90**	**0.007**	**2.74**	**0.008**	**38.91**	**0.47**	**0.09**	**0.88**	**20.01**

CFIA (Historical)		Al	As	Cd	Ca	Co	Cu	Pb	Mg	Mn	Se	Sr	Zn
Butter clams	***Mean***	21.31	2.62	0.13			1.88	0.079					14.27
	95% CI	6.13	0.89	0.03			0.19	0.022					0.78
Manila clams	***Mean***	33.86	3.01	0.44			1.62	0.121					14.21
	95% CI	7.53	0.49	0.09			0.17	0.050					0.99
Oysters	***Mean***	14.08	1.16	1.96			13.23	0.175					181.51
	95% CI	4.08	0.25	0.53			2.11	0.127					35.11
Scallops (whole)	***Mean***	11.75	0.41	2.25			1.35	0.030					47.32
	95% CI	5.93	0.18	1.23			0.64	0.009					22.44
Geoducks	***Mean***	11.09	2.29	0.19			3.21	0.053					20.60
	95% CI	4.69	0.88	0.07			0.74	0.025					3.18

The data reduction method accounted for 95.3% in the first two PCA axes (Figure 55) and clearly differentiated tissue on the basis of their respective trace metal loadings. The samples cluster in three distinct groups with no overlap that would indicate similarity among any of the tissue types in terms of trace metal composition. The adductor muscle samples are portrayed in gray, the gonad material in dark red, and the remaining viscera in green.

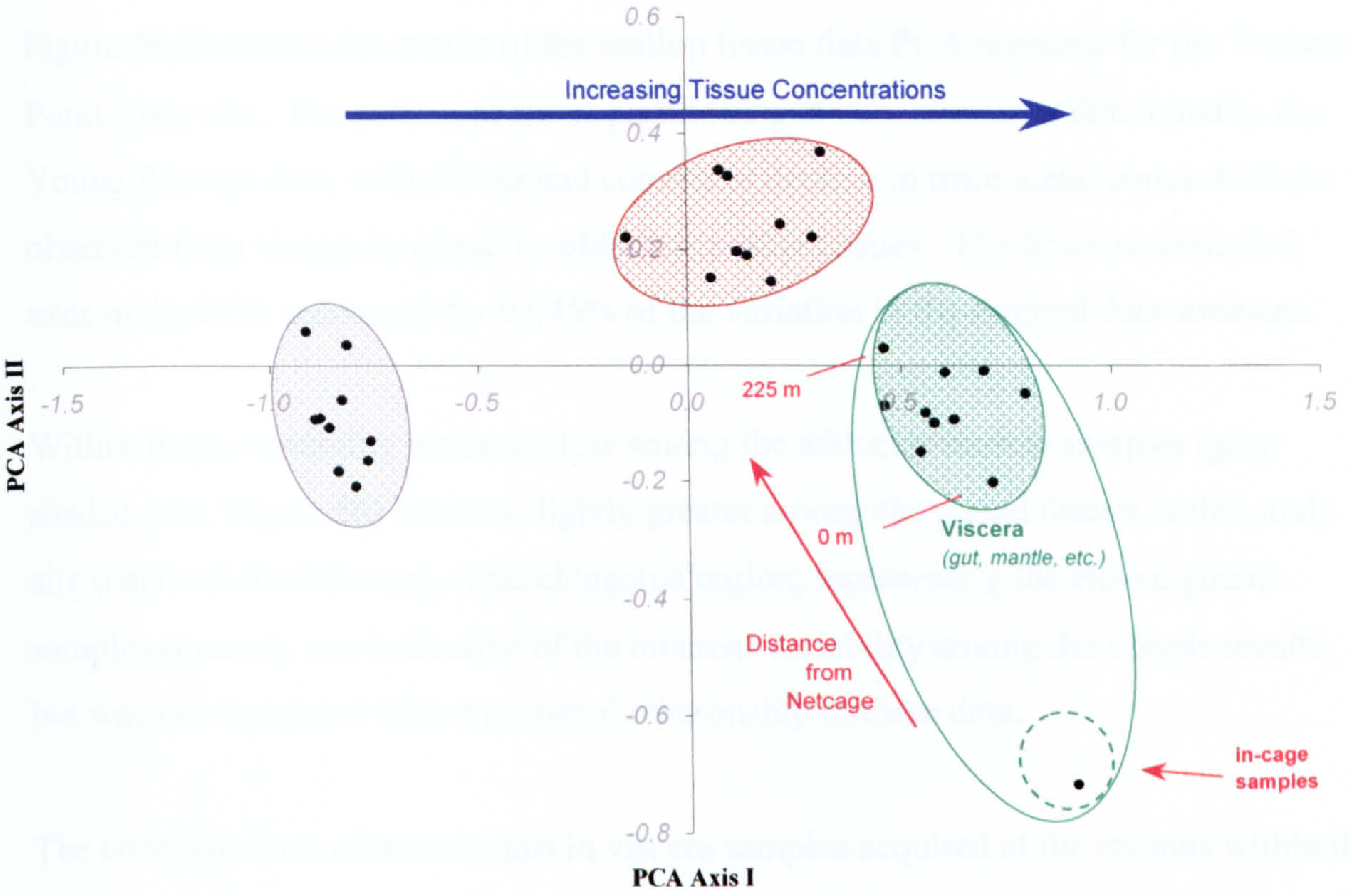
Given that close proximity of two samples in PCA space represents similarity, it is clear from Figure 55 that gonad and roe samples varied little (among the individual samples for each tissue type). In contrast, viscera samples suggested some spatial relationship in tissue levels of trace metals, with this difference related primarily to elevated levels found in scallops grown directly within the finfish cage system – the outlier shown in Figure 55. The remaining samples within this cluster also showed a minor gradient with distance from the cage system (indicated with red arrow; sample labels shown for 0 and 225-meter stations to indicate spatial relationship).

Tissue concentrations of trace metals increased from muscle to gonad to visceral tissues consistently for all constituents, summarized pictorially by the blue arrow on Figure 55. The relationship between these three tissue types was further explored with a partitioning ratio (lower table of Figure 55), which indicates the proportion of trace metals in each tissue standardized to adductor muscle levels.

The partitioning ratios calculated in Figure 55 revealed that visceral tissues included trace metal levels as high as 8.4 times that of the adductor muscle (copper concentrations) and that they were typically higher for all of these constituents. Manganese was the exception with slightly higher levels in the gonad tissue than in the remaining viscera. Other metals, such as magnesium and arsenic, maintained virtually identical levels among these two tissue categories, although each remained above that of the adductor muscle.

FIGURE 55

Trace metal partitioning in scallops (*Patinopecten yessoensis*) grown at Young Passage farm site. Centered Principal Components Analysis (PCA) of tissue samples (adductor muscle, roe, remaining viscera) acquired over study period (individual station means) using 13 trace metal constituents. PCA Axes I and II account for 95.27% of the variation within the original data structure.



	Adductor Muscle (ug/g)				Roe (ug/g)				Viscera (ug/g)				Levels elevated in Netcage*	Tissue Partitioning Ratio (viscera:roe:m
	Mean	*Stdev.*	*Max*	*Min*	*Mean*	*Stdev.*	*Max*	*Min*	*Mean*	*Stdev.*	*Max*	*Min*		
Al	5.26	1.60	8.00	2.29	23.63	2.39	27.00	20.67	26.06	2.57	29.50	22.71		5.0 : 4.5 : 1
As	0.93	0.10	1.07	0.72	0.92	0.06	1.00	0.85	1.08	0.05	1.15	0.98	1.45	1.2 : 1.0 : 1
Ba	0.037	0.01	0.05	0.01	0.20	0.03	0.24	0.14	0.23	0.02	0.25	0.20		6.2 : 5.3 : 1
Cd	1.62	0.39	2.16	0.93	3.70	1.08	5.55	2.21	5.55	0.40	6.29	4.93	7.61	3.4 : 2.3 : 1
Ca	126.70	19.08	157.75	100.50	298.28	39.81	383.33	253.33	412.13	41.29	511.38	359.14		3.3 : 2.4 : 1
Cu	0.38	0.04	0.46	0.32	1.85	0.22	2.18	1.44	3.18	0.50	4.31	2.65	6.23	8.4 : 4.9 : 1
Mg	500.98	15.27	519.67	480.00	811.45	33.65	857.50	733.00	807.57	26.93	855.00	774.22		1.6 : 1.6 : 1
Mn	1.29	0.45	2.45	0.86	5.12	1.05	6.71	3.67	4.25	0.55	5.50	3.61		3.3 : 4.0 : 1
Mo	0.02	0.00	0.02	0.01	0.05	0.01	0.06	0.04	0.09	0.00	0.10	0.08		5.3 : 3.1 : 1
Ni	0.43	0.07	0.53	0.35	1.11	0.16	1.33	0.83	1.74	0.11	1.99	1.59		4.1 : 2.6 : 1
Se	0.29	0.01	0.32	0.28	0.72	0.09	0.87	0.60	1.00	0.04	1.05	0.91	1.15	3.4 : 2.5 : 1
Sr	1.64	0.22	1.98	1.31	4.93	0.50	5.57	3.75	5.57	0.25	6.11	5.27		3.4 : 3.0 : 1
Zn	15.19	0.58	15.88	14.04	24.61	2.45	27.75	20.27	31.08	1.97	34.26	28.70	37.85	2.0 : 1.6 : 1

*** NOTE:** viscera tissue levels of trace metal sample constituents in excess of range at other stations.

The tissue levels in samples acquired from within the finfish cage system reported levels that were, for many of the trace metals, higher than those for any of the other downstream samples. In particular, cadmium, copper, zinc, arsenic, and selenium were found in levels slightly higher than the range of concentrations found in the other samples. Figure 55 (lower table) presents the average concentrations (with standard deviation), and concentration range, for each sampling station.

Figure 56 illustrates the results of the scallop tissue data PCA acquired for the Venture Point study site. The pattern of tissue partitioning is very similar to that found in the Young Passage data, with a clear and consistent decline in trace metal concentrations observed from viscera to gonad to adductor muscle tissues. The first two extracted axes in the PCA accounted for 95.45% of the variation in the original data structure.

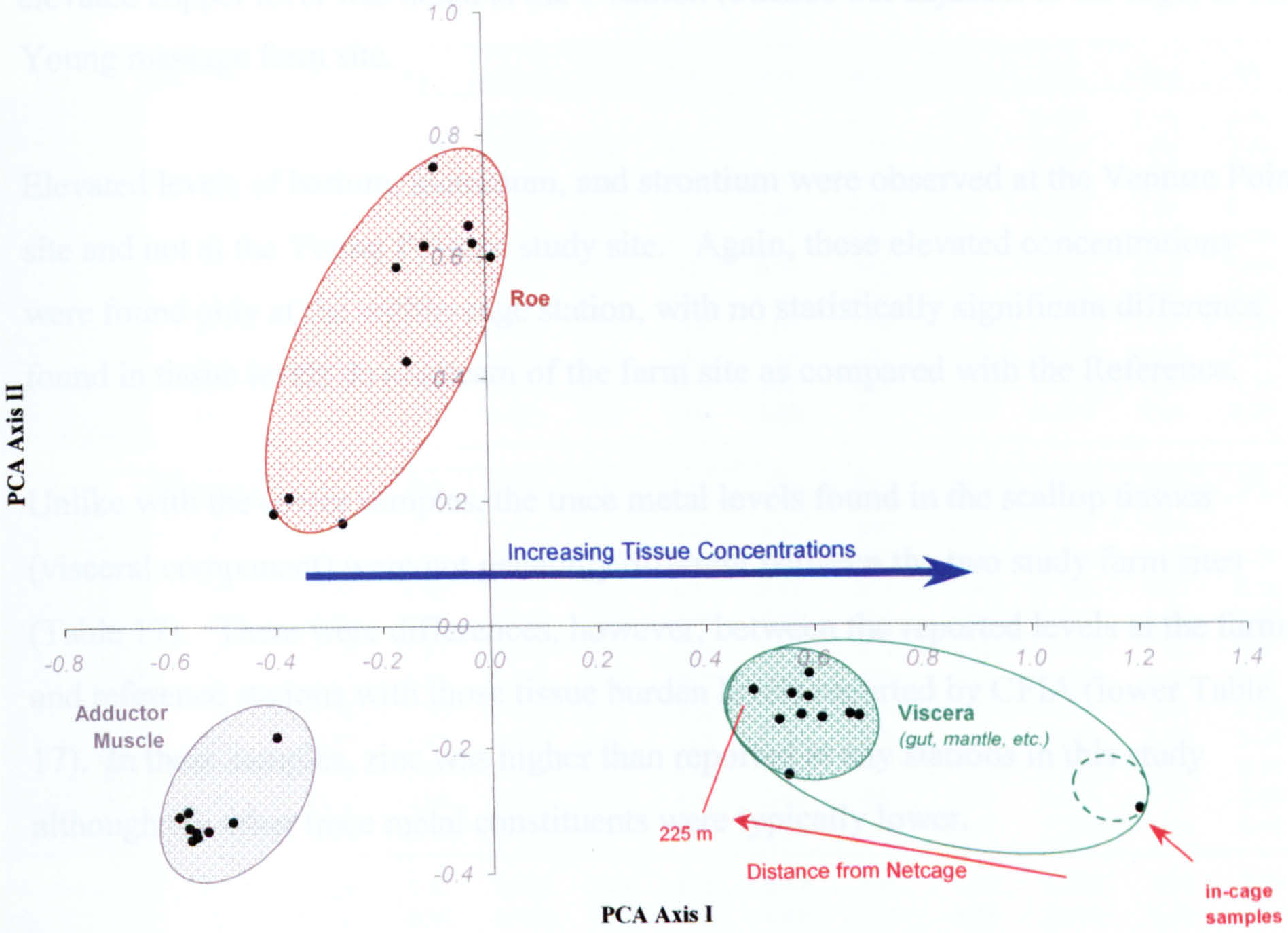
Within-tissue variability remained low among the adductor muscle samples (gray shaded area, Figure 56) but was slightly greater among the gonad tissues at this study site (dark red shaded area). This elongated region, representing the eleven gonad samples (means), was indicative of the inherent variability among the sample results, but was not correlated with any spatial relationship in these data.

The concentrations of trace metals in viscera samples acquired at the stations within the net cage system were also significantly higher than those at any of the downstream stations. Figure 56 shows the sample outlier within the viscera cluster, again supporting a spatial relationship (gradient) among these samples.

Table 17 summarizes the trace metal results acquired for the scallops (*Patinopectin yessoensis;* visceral tissue) sampled at the Young Passage (upper table) and Venture Point (lower table) study sites. These data represent sample means (n=7) for the 12 trace metal constituents that reported levels above their respective levels of analytical detection. Tissue levels of scallop reference samples are shown in blue, below the Venture Point data, and historical scallop tissue data collected by the Canadian Food Inspection Agency (CFIA) of wild-stock samples are presented at the bottom of this table.

FIGURE 56:

Trace metal partitioning in scallops (*Patinopecten yessoensis*) grown at Venture Point farm site. Centered Principal Components Analysis (PCA) of tissue samples (adductor muscle, roe, remaining viscera) acquired over study period (individual station means) using 13 trace metal constituents. PCA Axes I and II account for 94.45% of the variation within the original data structure.



	Adductor Muscle *(ug/g)*				**Roe** *(ug/g)*				**Viscera** *(ug/g)*				*Levels elevated in Netcage**	**Tissue Partitioning Ratio** *(viscera:roe:muscle)*
	Mean	*Stdev.*	*Max*	*Min*	*Mean*	*Stdev.*	*Max*	*Min*	*Mean*	*Stdev.*	*Max*	*Min*		
Al	4.03	1.44	7.83	2.83	17.35	4.67	24.00	9.50	28.81	2.24	33.00	25.29	61.00	7.1 : 4.3 : 1
As	0.77	0.04	0.82	0.69	0.53	0.05	0.61	0.44	0.95	0.05	1.01	0.83	1.56	1.2 : 0.7 : 1
Ba	0.029	0.01	0.06	0.02	0.13	0.04	0.18	0.07	0.29	0.02	0.33	0.27	0.54	9.9 : 4.4 : 1
Cd	0.95	0.46	2.02	0.41	2.42	0.88	3.64	0.66	5.59	0.38	6.21	4.98	8.15	5.9 : 2.6 : 1
Ca	129.75	23.21	184.33	109.33	420.85	156.13	814.00	252.00	452.32	38.27	504.43	388.57	590.00	3.5 : 3.2 : 1
Cu	0.37	0.11	0.66	0.31	1.09	0.28	1.45	0.56	2.47	0.23	2.80	2.17	3.87	6.8 : 3.0 : 1
Mg	499.81	38.16	601.50	472.50	969.75	129.92	1080.00	781.50	862.56	30.62	901.43	800.57		1.7 : 1.9 : 1
Mn	0.89	0.28	1.59	0.62	3.97	1.07	5.67	1.95	4.37	0.32	5.01	4.04		4.9 : 4.4 : 1
Mo	0.13	0.11	0.43	0.08	0.32	0.07	0.65	0.15	0.35	0.09	0.50	0.14		2.8 : 2.5 : 1
Ni	0.49	0.10	0.77	0.40	0.54	0.07	0.70	0.45	1.95	0.13	2.21	1.78		4.0 : 1.1 : 1
Se	0.69	0.30	1.51	0.53	5.85	1.63	7.85	3.50	1.63	0.21	1.85	1.11		3.4 : 8.5 : 1
Sr	1.60	0.13	1.84	1.44	1.26	0.19	1.80	1.14	5.99	0.34	6.60	5.48	7.14	3.7 : 0.8 : 1
Zn	14.81	0.77	16.80	14.12	13.89	0.79	15.10	12.90	32.51	4.62	37.90	25.74	38.90	2.2 : 0.9 : 1

*** NOTE:** viscera tissue levels of trace metal sample constituents in excess of range at other stations.

A statistical evaluation (t-tests), comparing downstream station values with reference samples, was conducted for each trace metal constituent. The highlighted (red) values in Table 17 indicate stations at which trace metal constituents were elevated above the levels reported at the Reference stations. Zinc, copper, calcium, cadmium and arsenic were elevated at both farm sites, although the higher concentrations occurred primarily at the station positioned directly within the finfish cage system (-15 meters). A single elevated copper level was noted at the 0-station (outside but adjacent to the cage) at the Young massage farm site.

Elevated levels of barium, aluminum, and strontium were observed at the Venture Point site and not at the Young Passage study site. Again, these elevated concentrations were found only at the within-cage station, with no statistically significant difference found in tissue levels downstream of the farm site as compared with the Reference.

Unlike with the oyster samples, the trace metal levels found in the scallop tissues (visceral component) were not generally different between the two study farm sites (Table 17). There were differences, however, between the reported levels at the farm and reference stations with those tissue burden levels reported by CFIA (lower Table 17). In these samples, zinc was higher than reported at any stations in this study although the other trace metal constituents were typically lower.

Further evaluation of the localized spatial trend in copper and zinc levels (near-field elevations) was completed in order to establish whether a temporal component to these tissue accumulations was apparent, and to further assess trends between the two study sites. Figure 57 provides a 3-dimensional contour plot for copper concentrations measured in scallop viscera over time (January/2002 through September/2003) and space (cage edge to 250 meters downstream of the farm). The upper plot illustrates these results for the Young Passage farm site and the lower plot portrays the same trends for Venture Point. The color scale has been standardized to allow direct comparisons of tissue concentrations within these two plots.

Table 17:

Mean trace metal (ug/g) tissue levels (n=7) for *Patinopecten yessoensis* (viscera) grown at Young Passage and Venture Point study sites. Reference values in blue, and historical shellfish data acquired for production and wild harvest areas of B.C. provided in bottom table (CFIA, 1999-2001; n=18). Values significantly greater than Reference shown in red.

Mean	Distance	Al	As	Ba	Cd	Ca	Cu	Mg	Mn	Mo	Se	Sr	Zn
Young Passage (Scallop Viscera)	-15	25.50	1.45	0.250	7.61	457	6.23	691	2.37	0.100	1.15	5.13	37.85
	0	27.10	1.14	0.236	6.29	415	4.31	778	4.12	0.094	1.00	5.48	34.26
	10	29.40	1.08	0.246	5.90	409	3.42	808	5.50	0.095	1.01	5.57	30.66
	20	29.38	1.10	0.249	5.55	511	3.64	855	4.74	0.093	1.05	6.11	32.81
	30	24.60	0.98	0.199	4.93	416	2.65	830	3.61	0.084	0.91	5.64	28.70
	50	24.50	1.10	0.214	5.79	432	3.28	843	4.09	0.085	1.05	5.85	30.84
	75	29.50	1.06	0.248	5.17	409	3.08	800	4.54	0.094	0.97	5.55	30.03
	100	22.71	1.15	0.201	5.78	359	2.81	804	4.15	0.089	1.01	5.35	31.00
	125	25.30	1.10	0.219	5.40	405	2.85	793	4.19	0.089	1.02	5.45	29.70
	175	23.88	1.08	0.229	5.56	398	2.97	792	3.91	0.084	1.04	5.48	33.93
	225	24.22	1.06	0.221	5.19	368	2.83	774	3.70	0.083	0.96	5.27	28.89

Mean	Distance	Al	As	Ba	Cd	Ca	Cu	Mg	Mn	Mo	Se	Sr	Zn
Venture Point (Scallop Viscera)	-15	61.00	1.56	0.540	8.15	590	3.87	848	4.34	0.110	1.20	7.14	38.90
	0	27.14	0.83	0.276	4.98	504	2.74	872	4.06	0.291	1.51	6.60	30.97
	10	25.29	0.92	0.266	5.54	439	2.77	875	4.26	0.353	1.58	6.14	33.72
	20	28.00	0.94	0.280	5.95	446	2.80	865	4.74	0.327	1.65	5.83	30.97
	30	29.00	0.97	0.300	5.47	389	2.47	846	4.20	0.369	1.66	5.54	29.82
	50	27.71	0.96	0.289	5.48	498	2.43	866	4.15	0.367	1.64	6.23	34.97
	75	29.29	0.98	0.281	5.19	402	2.17	801	4.04	0.140	1.11	5.82	37.68
	100	31.86	1.01	0.333	5.84	471	2.39	876	4.54	0.384	1.83	6.06	37.55
	125	27.86	0.96	0.293	6.21	480	2.40	901	4.25	0.400	1.85	6.23	37.90
	175	33.00	0.99	0.309	5.93	433	2.30	897	5.01	0.501	1.77	5.94	25.78
	225	29.00	0.95	0.281	5.33	461	2.20	828	4.47	0.393	1.65	5.48	25.74
Reference	Mean:	28.06	1.02	0.265	5.48	413	2.59	819	4.23	0.252	1.35	5.51	28.88
	Stdev	16.42	0.31	0.120	1.76	125	0.94	130	1.64	0.671	1.45	1.10	5.91
	n	31	32	31	32	32	32	32	32	32	31	30	30
	95 CL	5.78	0.11	0.042	0.61	43	0.33	45	0.57	0.232	0.51	0.39	2.11

CFIA (Historical)		Al	As	Ba	Cd	Ca	Cu	Mg	Mn	Mo	Se	Sr	Zn
Butter clams	*Mean*	21.31	2.62		0.13		1.88						14.27
	95% CI	6.13	0.89		0.03		0.19						0.78
Manila clams	*Mean*	33.86	3.01		0.44		1.62						14.21
	95% CI	7.53	0.49		0.09		0.17						0.99
Oysters	*Mean*	14.08	1.16		1.96		13.23						181.51
	95% CI	4.08	0.25		0.53		2.11						35.11
Scallops (whole)	*Mean*	11.75	0.41		2.25		1.35						47.32
	95% CI	5.93	0.18		1.23		0.64						22.44
Scallops (muscle)		5.17	0.60		0.42		0.46						18.47
Geoducks	*Mean*	11.09	2.29		0.19		3.21						20.60
	95% CI	4.69	0.88		0.07		0.74						3.18

Figure 57 indicates a clear temporal variation in scallop tissue copper concentrations for both the Young Passage and Venture Point farm sites. The Young Passage site (upper plot) suggests that copper concentrations remain at low levels (< 2 ug/g) during the winter period, January through early March (Figure 57 – A), and increase to levels of 4-6 ug/g that persist for the remainder of the year (Figure 57 – B). There did not appear to be any distinct spatial trend within these data, with tissue levels remaining at comparable levels across all stations downstream of the finfish system.

The spatial and temporal pattern in tissue copper concentrations for the Venture Point site were completely different than that portrayed for the Young Passage site. At this site (Figure 57, lower plot) the tissue levels for copper appeared to remain low throughout most of the year, with values generally < 2 ug/g found in the scallop viscera samples. Although there was no distinct seasonal fluctuation noted in these results, a short period of elevated tissue levels occurred during January/2003 (Figure 57 – C).

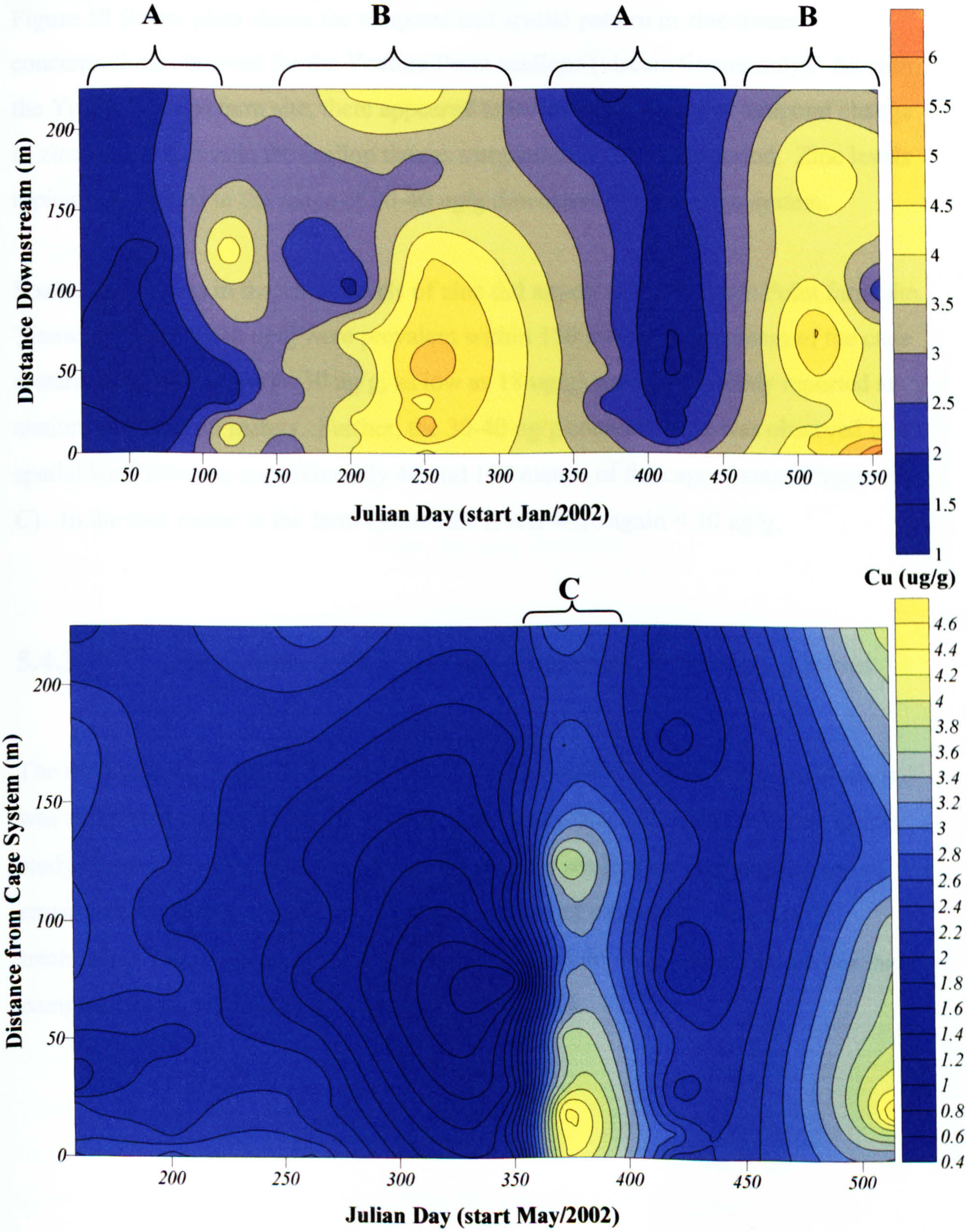
The scallop tissue levels during this period (Figure 57 – C) also appeared to include a spatial component, with concentrations decreasing with distance of the farm and occurring primarily within the near-field region (< 50 meters downstream of the cage system). This short-term copper "spike" coincided with the installation of new, copper-treated nets at this farm site that occurred at the onset of the new finfish production cycle (January/2003 – see Figure 38).

The fluctuations in copper levels observed in these scallop tissues were not correlated with farm site production, feeding rate and thus with the temporal change in waste flux for either of the study sites.

The temporal and spatial pattern in tissue levels of zinc was assessed in a similar manner as that for copper. Figure 58 presents a 3-dimensional color contour plot for the Young Passage (upper plot) and the Venture Point zinc data (scallop viscera only). The spatial data reflect tissue concentration from the edge of cage downstream to 225 meters, with the in-cage results removed from this analysis to clarify patterns outside of the system.

Figure 57:

Spatial and temporal changes in copper levels in scallop viscera acquired for the Young Passage farm site (upper plot), January/2002 through September/2003, and for Venture Point site (lower plot) from May/2002 through September/2003. Data as ug/g tissue (viscera only: roe and adductor muscle excluded).



At the Young Passage farm site zinc concentrations appeared to be lower during the initial sampling periods (Figure 58 – A), with tissue levels generally < 30 ug/g. Over the remainder of the study (May/2002 through September/2003) the zinc concentrations were typically in the range of 30-40 ug/g (Figure 58 – B). There did not appear to be any clear downstream pattern in tissue levels at this farm site.

Figure 58 (lower plot) shows the temporal and spatial pattern in zinc tissue concentrations observed for the Venture Point scallops (viscera tissues only). As with the Young Passage farm site, there appeared to be little in the way of temporal change in zinc concentration in the scallop tissues sampled over the study period. Zinc levels typically remained in the range of 30-40 ug/g downstream of the cage system.

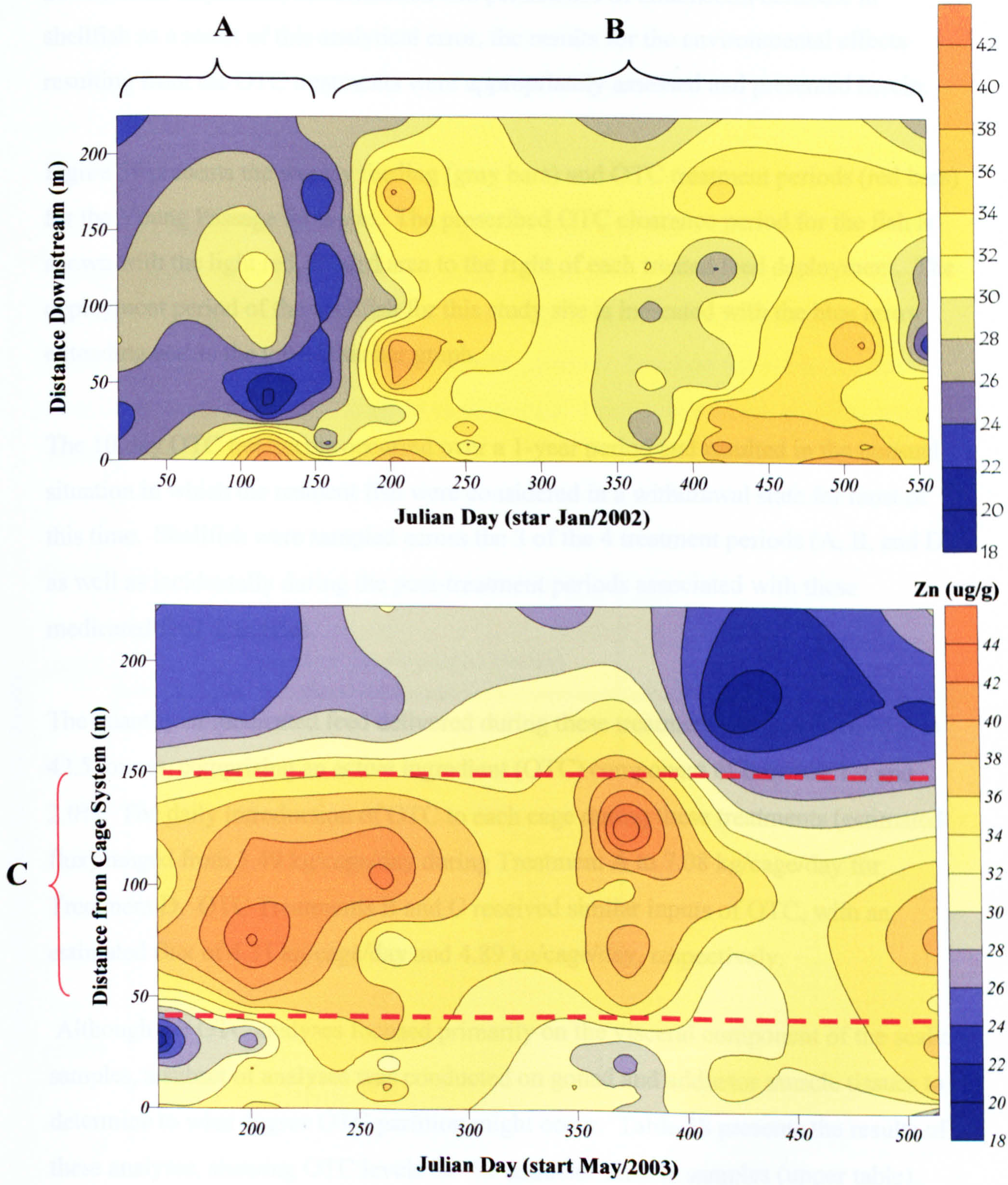
Spatial differences in the tissue levels of zinc did appear at the Venture Point farm site. Tissue levels of 30-40 ug/g were prevalent within 150 meters downstream of the cage system, but lower levels (< 30 ug/g; as low as 18 ug/g) were consistently reported at stations beyond 150 meters. Further, the 30-40 ug/g contour range was observed in a spatial band between approximately 40 and 150 meters of the cage system (Figure 58 – C). In the area closer to the farm system the levels were again < 30 ug/g.

5.4.3 Antibiotics and Chemotherapeutics in Scallop Tissues

The evaluation of shellfish for tissue accumulation of chemotherapeutic compounds was completed opportunistically at each of the study sites. Over the duration of this study, Venture Point received a single treatment for sea lice (in-feed application of emamectin benzoate) and Young Passage a number of oxytetracycline (OTC) treatments. The respective treatment chronology and delivery specifications for these compounds was presented in Chapter 4, Section 4.4.4.

Figure 58:

Spatial and temporal changes in zinc levels in scallop viscera acquired for the Young Passage farm site (upper plot), January/2002 through September/2003, and Venture Point farm site (lower plot) from May/2002 through September/2003. Data as ug/g tissue (viscera only: roe and adductor muscle excluded).



Although scallop tissue samples for emamectin benzoate analysis were acquired post-treatment, an undetected error in establishing GC-MS instrument calibration and standardization resulted in the use of an inappropriate minimal detection limit (0.5 ug/g rather than ug/kg). The resulting data all reported tissue concentrations of < 0.5 ug/g despite the expectation of some detectable near-field accumulation of this compound and/or its metabolite residues. Despite the missed opportunity to evaluate the downstream dispersion, accumulation and persistence of emamectin benzoate in shellfish as a result of this analytical error, the results for the environmental effects resulting from the OTC treatments were appropriately assessed and presented herein.

Figure 59 presents the weekly feeding (gray bars) and OTC treatment periods (red bars) for the Young Passage farm site. The prescribed OTC clearance period for the fish is shown with the light red, shaded area to the right of each treated feed deployment. The deployment period of the shellfish for this study site is indicated with the blue arrow extending across the top of this bar graph.

The 10-day OTC treatments occurred over a 1-year period and resulted in the unusual situation in which the resident fish were considered in a withdrawal state for most of this time. Shellfish were sampled across the 3 of the 4 treatment periods (A, B, and D), as well as incidentally during the post-treatment periods associated with these medicated feed deliveries.

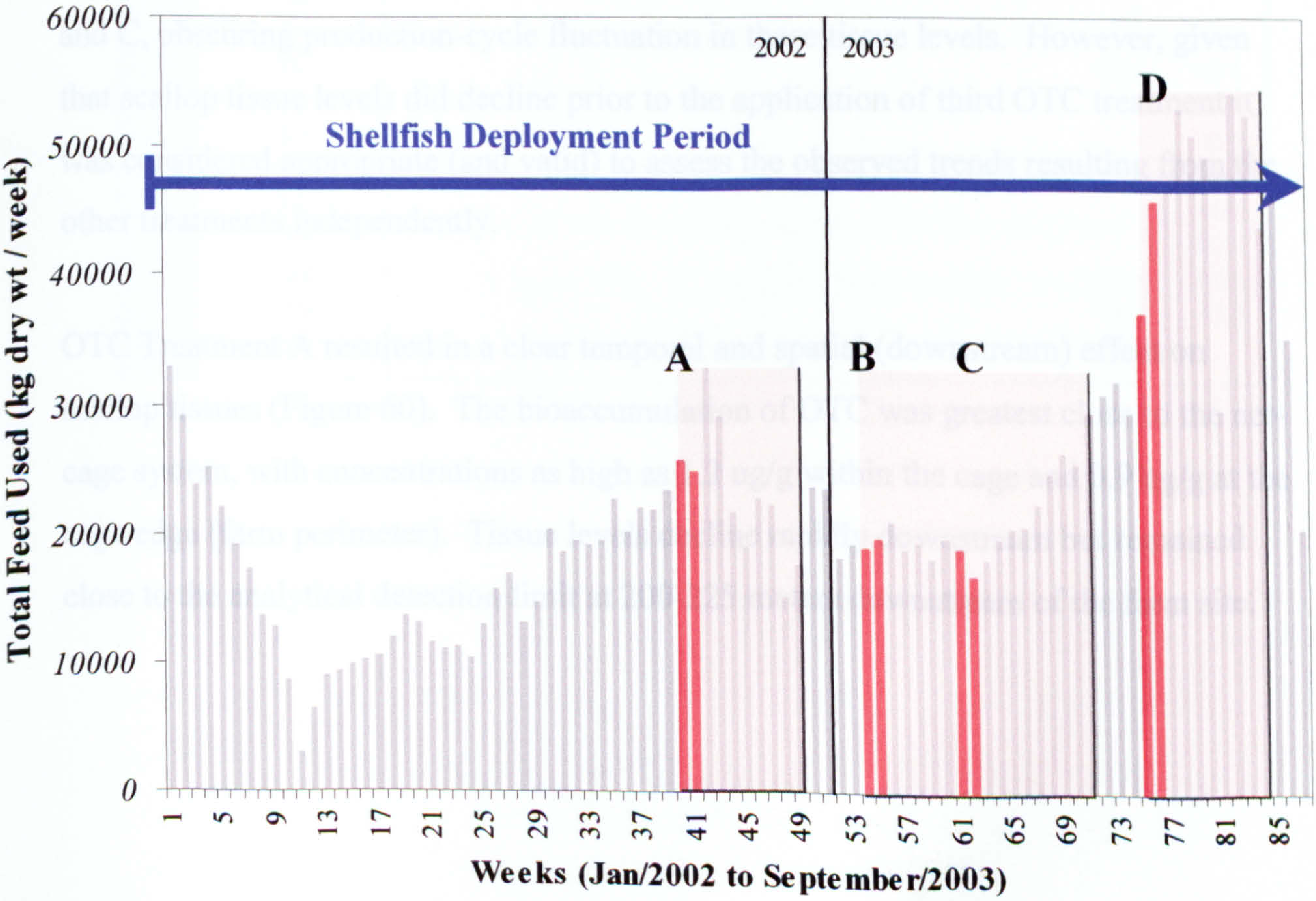
The quantity of medicated feed delivered during these treatments ranged from 35.2 to 42.5 tonnes, comprising an active ingredient (OTC) component of between 1.2 and 2.0%. The daily introduction of OTC to each cage during these treatments (estimated flux) ranged from 3.49 kg/cage/day during Treatment A to 7.08 kg/cage/day for Treatment D. OTC Treatments B and C received similar inputs of OTC, with an estimated flux of 4.31 kg/cage/day and 4.89 kg/cage/day, respectively.

Although the OTC analyses focused primarily on the visceral component of the scallop samples, a subset of analyses was conducted on gonad and adductor muscle tissues to determine to what degree OTC partition might occur. Table 18 presents the results of these analyses, showing OTC levels for the adductor muscle samples (upper table) taken on 9 site surveys of the Young Passage farm site. Samples taken at each of the downstream stations were assessed for gonad concentrations of oxytetracycline residues on only two of these occasions (lower portion of Table 18).

Analysis of adductor muscle samples confirmed that no oxytetracycline, or tetracycline, residues accumulated in these tissues during or following farm treatment with this antibiotic compound (Table 18). Concentrations remained below the analytical limits of detection established for these analyses (0.1 ug/g).

Figure 59:

Weekly feed input (kg) to Young Passage farm site (gray bars) showing oxytetracycline (OTC) treatment (red bars) and prescribed withdrawal periods (light red shaded areas). Each of the four required treatments (A-D) was delivered with medicated feed (milled) over a 10-day feeding period. Tonnage fed out to the fish over these treatments varied from 35 to 42 tonnes. Period over which shellfish were monitored is shown with a blue arrow.



Analysis of the gonad tissue samples indicated that OTC could be detected in these tissues, although concentrations never exceeded 0.3 ug/g (Table 18). Accumulation occurred during the January/2003 antibiotic treatment and subsequent sampling in February suggested that the OTC had disappeared.

Downstream dispersion of the OTC residues, and near-field bioavailability to the study scallops, was also supported in these data. Gonad tissue levels of OTC were present at levels above analytical detection at stations from in-cage (-15 meter) and downstream to 100 meters (values indicated in red in Table 18). There was no indication of persistence of these residues beyond this distance.

The accumulation of oxytetracycline residues in scallop viscera was assessed primarily during treatment periods A and D. A single sample was taken during Treatment B and a number of incidental samples were processed between treatments to establish tissue levels in the test animals.

Figure 60 provides a 3-dimensional contour plot of these data to illustrate the spatial and temporal pattern of OTC uptake in the study scallops at the Young Passage farm site. A tissue concentration gap is probable due to missing data during Treatments B and C, obscuring production-cycle fluctuation in these tissue levels. However, given that scallop tissue levels did decline prior to the application of third OTC treatment it was considered appropriate (and valid) to assess the observed trends resulting from the other treatments independently.

OTC Treatment A resulted in a clear temporal and spatial (downstream) effect on scallop tissues (Figure 60). The bioaccumulation of OTC was greatest close to the net-cage system, with concentrations as high as 1.2 ug/g within the cage and 0.9 ug/g at the cage edge (farm perimeter). Tissue levels decline rapidly downstream but remained close to the analytical detection limit at 200-225 meters downstream of the farm site.

Table 18:

OTC residues (ug/g) found in scallop adductor muscle and gonad tissue collected at the Young Passage farm site during the study period. Analytical detection limit of 0.1 ug/u was used for all analyses. Tissues burden levels in excess of the detection limit were shown in red.

Scallop Adductor Muscle OTC Residue Distance from Netcage (m)	18-Jan-02	12-Mar-02	2-Oct-02	5-Oct-02	8-Oct-02	11-Oct-02	14-Oct-02	6-Jan-03	21-Feb-03
-15	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
0	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
10	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
20	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
30	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
50	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
75	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
100	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
125	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
175	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
225	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Scallop Gonad Tissue OTC Residue Distance from Netcage (m)	6-Jan-03	21-Feb-03
-15		
0	0.2	<0.1
10	<0.1	<0.1
20	0.3	<0.1
30	0.2	<0.1
50	0.3	<0.1
75	0.2	<0.1
100	0.2	<0.1
125	<0.1	<0.1
175	<0.1	<0.1
225	<0.1	

The scallops appeared to accumulate OTC residues immediately upon initiation of the first medicated feed treatment in October/2002, with increasing levels developing during this period and within the post-treatment period (Figure 60 - A). Extrapolation from the contour plot suggests that peak tissue levels occurred within 16 days of the start of treatment, indicating that scallops acquired OTC residues during the 10-day treatment period as well as 1 week beyond the delivery of the medicated feed (initial period of OTC clearance for the fish). An uptake rate of 0.056 ug/g/day was estimated using these concentration data.

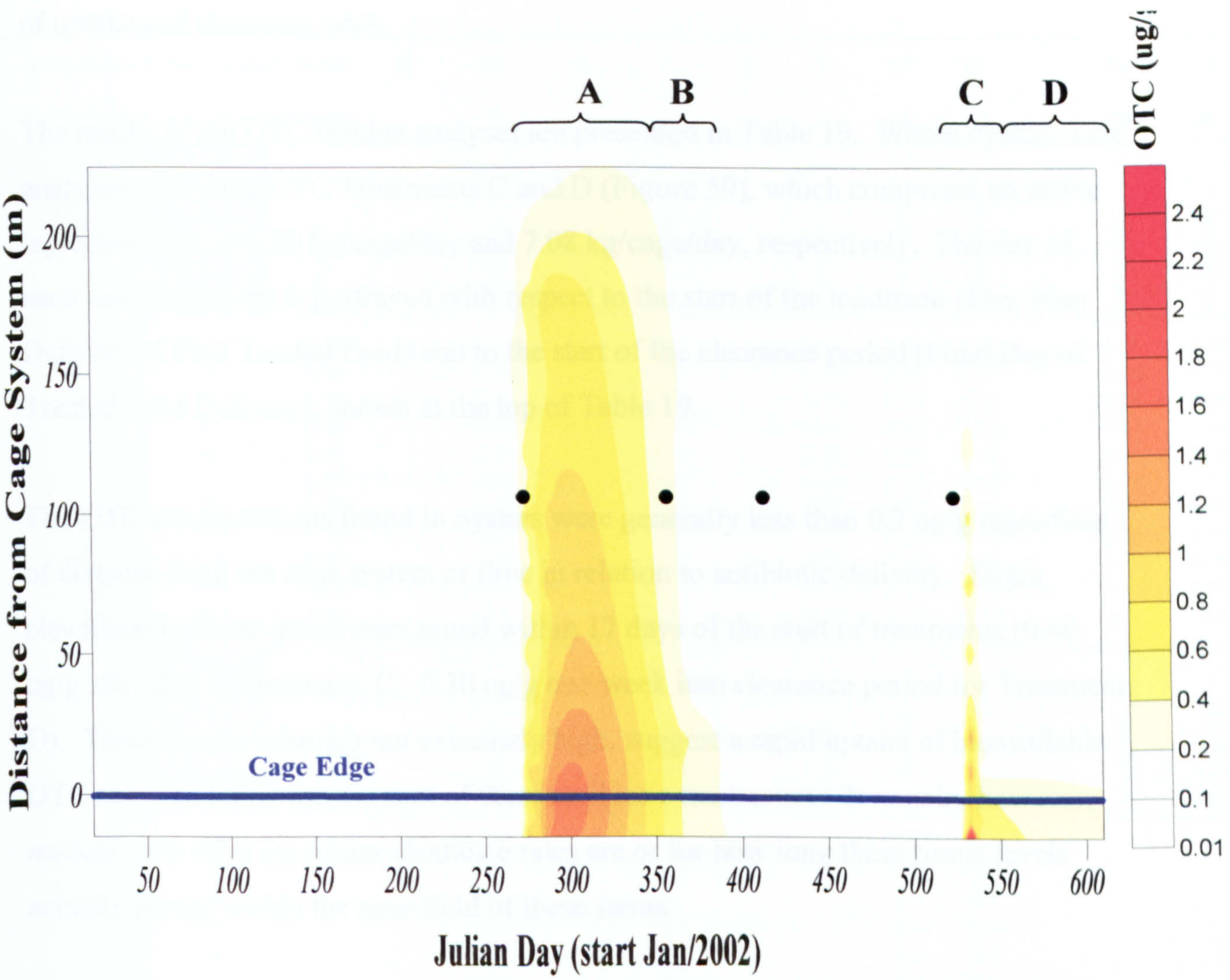
Loss of OTC residues from the scallops occurred considerably slower than the estimated uptake rate. Although the second OTC treatment occurred shortly after the first, an estimate for OTC clearance of 0.016 ug/g/day was derived from these data.

The application of OTC during Treatment D occurred in June/2003 and resulted in an OTC flux of over twice that of Treatment A (3.49 kg/cage/day during Treatment A and 7.08 kg/cage/day for Treatment D). Scallop viscera levels, however, suggested only minor differences at the net-cage edge with tissue levels of 1.7 ug/g reported for the net cage edge (station 0), opposed to the 1.4 ug/g derived from the Treatment A tissue analyses. An in-cage sample taken during this latter Treatment revealed scallop tissue levels of 2.7 ug/g, representing an approximately 37% decline across the net and downstream of the cage system.

The temporal fluctuation in OTC levels in scallop tissues was also substantially different during Treatment D (Figure 60). A rapid increase in OTC levels, reaching the reported in-cage levels in 13 days following initiation of the in-feed treatment, appeared to be followed by a tissue clearance rate (0.109 ug/g/day) over 10-fold of that estimated for the first OTC treatment applied in October/2002. Some persistence in OTC tissue levels occurred within the near-field stations (< 10 meters) following clearance at all downstream stations (Figure 60 – D).

Figure 60:

3-dimensional contour plot of oxytetracycline (OTC) concentration data acquired for scallop viscera (excluding roe and adductor muscle) at Young Passage farm site from January/2002 through September/2003. Start of each treatment shown with black dot. **A:** tissue levels during and following the 10-day Treatment A initiated in October/2002. B: some near-field persistence in OTC tissue residues due to Treatment B application in January/2003. **C:** similar OTC treatment applied in June/2003. **D:** persistence of OTC tissue residues in near-field following June/2003 treatment.



The dispersion pattern in scallop OTC tissue levels indicated a temporally narrow band that extended downstream of the farm site to approximately 125 meters. Tissue concentrations declined along this downstream transect, but remained at levels < 0.4 ug/g. Significantly elevated OTC levels were constrained to the area within and directly adjacent to the finfish net-cage system, as indicated above.

5.4.4 Antibiotics and Chemotherapeutics in Oyster Tissues

Interpretation of the analysis results of whole oyster tissues for emamectin benzoate residues was constrained by the same analytical error identified for the scallop component of this study (see Section 5.4.3). In addition, deployment of oysters occurred later in the study with much fewer test animals than anticipated would be necessary for all components of the study, and as such the frequency of sample acquisition for the antibiotic residue analyses were less than optimal to allow estimation of uptake and clearance rates.

The results of the OTC residue analyses are presented in Table 19. Whole oysters were analyzed following OTC Treatments C and D (Figure 59), which comprised an active ingredient flux of 4.89 kg/cage/day and 7.08 kg/cage/day, respectively. The day of each sampling event is portrayed with respect to the start of the treatment (Day from Delivery of First Treated Feed) and to the start of the clearance period (Final Day of Treated Feed Delivery), shown at the top of Table 19.

The OTC concentrations found in oysters were generally less than 0.2 ug/g regardless of distance from the cage system or time in relation to antibiotic delivery. Slight elevations in tissue levels were noted within 17 days of the start of treatments (0.40 ug/g after day 1, Treatment C; 0.30 ug/g one week into clearance period for Treatment D). These levels, although not extremely high, suggest a rapid uptake of bioavailable OTC residues following the start of these antibiotic treatments. It remain, however, unclear as to what the oyster clearance rates are or for how long these tissue levels actually persist within the near-field of these farms.

Both oxytetracycline and tetracycline residues were analysed in the collected oyster tissue samples. While the above trends apply to the former antibiotic residue, there were no tetracycline levels detected in any of the samples submitted for analysis.

Table 19:

Oxytetracycline (OTC) and tetracycline (TC) concentration data (ug/g) acquired for whole oysters at Young Passage farm site from January/2003 through September/2003. Start of each OTC treatment indicated with red vertical bars. Days from the start of each treatment, and the days from end of treatment (beginning of post-treatment, or clearance period) indicated at the top of the table.

Days from First Treated Feed:			0	26	0	1	0	2	17	21	59	70
Days from Last Treated Feed:				15				-8	7	11	49	60
	Whole Oyster Tissue OTC Residue	9-Jan-03	Treatment B	5-Feb-03	Treatment C	28-Feb-03	Treatment D	5-Jun-03	20-Jun-03	24-Jun-03	1-Aug-03	12-Aug-03
Distance from Netcage (m)	**-15**	0.10		0.15		0.40		0.10	0.30	0.10	0.15	0.10
	0	0.10		0.18		0.40		0.10	0.20	0.10	0.14	0.16
	10	0.10		0.17		0.30		0.10	0.20	0.10	0.10	0.20
	20	0.10		0.12		0.40		0.10	0.20	0.10	0.15	0.18
	30	0.10		0.15		0.30		0.10	0.20	0.10	0.11	0.16
	50	0.10		0.13		0.30		0.10	0.20	0.10	0.10	0.12
	75	0.10		0.18		0.30		0.10	0.20	0.10	0.10	0.15
	100	0.10		0.14		0.30		0.10	0.20	0.10	0.10	0.14
	125	0.10		0.12		0.30		0.10	0.10	0.10	0.13	0.16
	175	0.10		0.13		0.20		0.10	0.10	0.10	0.11	0.11
	225	0.10		0.1		0.10		0.10	0.10	0.10	0.1	0.1
	Whole Oyster Tissue TC Residue	9-Jan-03		5-Feb-03		28-Feb-03		5-Jun-03	20-Jun-03	24-Jun-03	1-Aug-03	12-Aug-03
Distance from Netcage (m)	**-15**			0.1		0.1			0.1	0.1		
	0			0.1		0.1			0.1	0.1		
	10			0.1		0.1			0.1	0.1		
	20			0.1		0.1			0.1	0.1		
	30			0.1		0.1			0.1	0.1		
	50			0.1		0.1			0.1	0.1		
	75			0.1		0.1			0.1	0.1		
	100			0.1		0.1			0.1	0.1		
	125			0.1		0.1			0.1	0.1		
	175			0.1		0.1			0.1	0.1		
	225			0.1		0.1			0.1	0.1		

5.4.5 Hydrocarbon Accumulation in Shellfish Tissue

The use of fuels (diesel, gas, oils) is routine on finfish farms. However, there is not typically a release of such materials except through an operational accident. A series of tissue samples were acquired and analyzed for evidence of release to the receiving environment. Of the 6 surveys that included this analytical component, none of the 250 samples indicated any accumulation of hydrocarbons (<50 ppm).

5.5 Discussion

The loss of contaminants from marine net-cage facilities, to the water column, occurs through a process of continual, low-level release over a production cycle (18-24 months), as well as through distinct, short-term pulses that can be measured in days. The former process involves the release of miconutrients (trace metals), a proportion of the introduced feed (Lorentzen and Maage, 1999; Lall, 1991; Tacon and De Silva, 1983), which are lost to the environment as faeces and urine. The latter includes the categories of antibiotic and chemotherapeutic (e.g., pesticide) compounds that are introduced as medicated feed for short-term treatment, and result in a similar loss to the environment in association with organic solids (faeces) and via a significant, soluble fraction (see Chapter 4).

The persistence of these waterborne contaminants, determined through the physiographic and oceanographic characteristics of a farm site (Chapter 3), will determine whether *in situ* concentrations of these waste constituents are of sufficient magnitude as to negatively affect non-target species located downstream of such facilities. Given the inherent filtration capacity of bivalve mollusks (Cole et al., 1992; Cranford et al., 1998), and their commercial fisheries and aquaculture importance, the evaluation of contaminant uptake and persistence in shellfish was considered an important component in the evaluation of Multi-Trophic Aquaculture (MTA), particularly from a seafood safety perspective.

5.5.1 Shellfish Tissue Quality and Trace Metals

This study component revealed that trace metal constituents of the feed did become available to shellfish, although the quantifiable accumulation of trace metals in these non-target shellfish species occurred only in close proximity to the cage system and only for the tested scallops (*Patinopectin yessoensis*). The levels found in the oyster (*Crassostrea gigas*) remained below, or equivalent to reference levels in all tissue samples analyzed.

Trace metal concentrations within the scallops were significantly higher at the in-cage station (shellfish suspended among the fish), as compared with reference levels, suggesting that the *in situ* concentrations of these metals may have been sufficient to elicit a measurable accumulation response in close proximity to the discharge source (the fish). The trace metal constituents found in the suspended shellfish positioned downstream of cage system did not report any significant accumulations above that of background. These results could be explained through a number of theoretical processes, although each of the proposed scenarios would require additional research to validate.

1. The dispersion of trace metals outside of the cage system occurs primarily within a settleable waste fraction that is dissipated below the portion of the water column supporting the shellfish.
2. Trace metal loadings within the suspended field are of insufficient magnitude (concentration) to elicit a significant downstream accumulation response in the scallops, primarily due to the physical processes that dissipate these materials.
3. The majority of the waterborne trace metals are bound within a fine particulate waste fraction that are selectively avoided by the shellfish in favour of a preferred seston diet.

The process suggested by the first scenario expands upon the waste dispersion process described in Chapter 4 and speculates that trace metal constituents of fish waste may adhere primarily to settleable organics and that the fate of this material is determined by settling rates (Chen et al., 1999), flocculation (Milligan et al., 2001), and site-specific oceanographic characteristics that influence these particle dynamics. Tidal quiescence and infrastructure interference may contribute to the localized distribution of these solids and, as shown in Chapter 4, result in a significant (if not the majority) of these wastes being discharged through the bottom of the net-cage system. If trace metals are dispersed within this particular waste pathway, the majority (if not all) of the available trace metals will occur below the water column layer that supports suspended shellfish aquaculture systems.

Although a trace metal waste signal was *detectable* within the net-cage system of the farm sites, the observed tissue concentrations were not at exceedingly high levels. Zinc, for example, revealed in-cage levels that were statistically greater than the reference (Young Passage site), but both of these values were lower than those reported by the CFIA in their historical seafood database.

The second scenario presented to explain these spatial patterns simply suggests that the flux of trace metals to the water column is insufficient to result in environmental concentrations that would facilitate biomagnification of these constituents within the shellfish tissues. Release of the initially low concentrations from within the net-cage (contaminant source), represented as a dissolved contaminant component or within an organic or inorganic matrix as fine particulate matter, the downstream dispersion and dilution of these waste constituents occurs rapidly and results in waterborne levels that become indistinguishable from background.

The flux of these trace metals to the water column (estimated for total zinc at 12.9% of the initial feed application – see Chapter 4), is likely represented by a combination of dissolved and fine (non-settleable) particulate fractions. Given these principle dispersion pathways it would appear unlikely that either would result in a distinct lack of tissue effects on the adjacent shellfish resources. Rapid uptake of a dissolved fraction by phytoplankton would be conveyed, indirectly, to the shellfish and presumably result in some level of bioaccumulation. In the trace metal constituents of the waste are dispersed in association with a fine particulate fraction (organic or inorganic matrix) it is intuitive that these materials could be directly ingested by shellfish and thereby result in elevated tissue levels of the various metals.

It has been speculated that most nonsiphonate shellfish species actively select their food from the seston fractions (Coma et al., 2001). Scallops have been shown to have the ability to select particles from natural assemblages of seston, and to preferentially reject particles of poorer quality (Bacon et al., 1998; MacDonald and Ward, 1994; Wong and Cheung, 1999). In addition, these studies revealed that particle selection efficiency increased with POM concentration, and that for the sea scallop (*Placopecten magellanicus*) a reduction in clearance rate and production of pseudofaeces was used to regulate ingestion when particulate levels increased.

Many studies have examined the effects of increasing particulate concentration on feeding behaviour in bivalves, including scallops, mussels, oysters and clams (MacDonald and Ward, 1994; Bacon et al., 1999; Cranford and Gant, 1990). If the fine particulate fraction of the organic wastes released from finfish farms is actively avoided by scallops (and other shellfish), and this fraction represents a primary sink for these trace metals, tissue level accumulation may be determined by the contribution of such materials within the natural seston field. Periods of natural food limitation (site-specific, seasonal) could also affect whether such materials continue to be avoided, or whether such conditions would dictate a need for these waste particles to be consumed.

Despite the process by which trace metals are released and dispersed within the water column at farm sites, the tissue burden effects on shellfish reported in this study were minimal and in fact well within natural levels for coastal British Columbia. The signal of farm-derived trace metals was generally weak, although particular metals showed some spatial and temporal trends that supported the contribution of this source to the water column levels of these natural occurring elements.

Copper and zinc were the primary trace metal constituents found at elevated levels within the scallops. Others included cadmium, selenium, calcium, barium and aluminum, all of which were identified as trace metals characteristic of the feed and the discharged waste material (Chapter 4). Partitioning of these metals within the scallop was clearly evident, with the highest levels observed in the viscera (without gonad, adductor muscle) and is presumed to primarily reflect the levels in the gut.

Copper and zinc accumulated in scallop tissues and achieved statistically significant levels at the in-cage station (discussed above). Further examination of the spatial and temporal change in these trace metal constituents suggested significant site-specific differences in natural background levels, considerable spatial variability, as well as compounding seasonal and contaminant source effects.

At the Young Passage farm site copper levels in scallop tissues revealed a distinct seasonal fluctuation. Levels in the winter and through early spring were generally < 2 ug/g but for the remainder of the year increased to levels of 4-6 ug/g. A lack in spatial pattern to these changes, or a correlation with farm activities (feeding regime, net changes, etc.), would suggest that these fluctuations were a result of natural inputs of copper to the water column. Given that this site is situated in a relatively quiet embayment, input of trace metals such as copper via terrestrial sources may result in localized persistence at levels that will be reflected in shellfish accumulations.

Copper levels in scallop tissues remained < 2.0 ug/g throughout the year at the other study farm site. Higher tidal flows and the potential lack of terrigenous sources for such trace metal influences may explain the natural differences between the two study sites. An introduction of copper oxide treated nets in early January/2003, at the beginning of a new fish production cycle at this farm site, did result in a measurable and localized tissue effect. Copper levels in the scallop viscera increased to above 4.5 ug/g following this introduction, but levels returned to background (< 2.0 ug/g) within 3 weeks. This downstream effect was observed to approximately 50 meters.

The release of copper from treated net-cages has been shown to contribute an estimated 0.16-0.18 ug/L to the water column above that of the background levels (Brooks, 2001). These values were estimated for conditions within the net-cage, and immediately downstream of the cage, but can not adequately account for the tidal mixing and dilution processes that will affect these copper fluxes and resulting bioavailable concentrations along the dispersion pathway.

Zinc is an important additive to fish feed and serves not only as an essential metalloenzyme (Desilva and Anderson, 1995) but it also reduces the risk of inducing cataracts in juvenile salmon (Richardson et al., 1986). Examination of the spatial and temporal patterns of zinc accumulation in scallop tissues, although levels were not statistically higher than those reported for reference conditions (or in historical CFIA seafood database), provided some insight into the natural processes affecting shellfish tissues levels of this trace metal.

Tissue levels were lower at the start of the monitoring program at the Young Passage farm site, perhaps an indication that the source of these test animals comprised growing waters with naturally lower levels of zinc. Once acclimated to the study site tissue levels achieved concentrations of 30-40 ug/g and remained generally constant over the two years of the study. There was, however, no apparent spatial trend that could be attributed to a farm source of zinc.

At the Venture Point site the variation in tissue levels of zinc was readily apparent from the spatial and temporal analysis. However, despite levels that remained below reference conditions and within the range typical of shellfish tissues collected from along the British Columbia coast, the analysis did suggest a very slight spatial relationship in these data. Stations between 50 and 150 meters downstream of the farm appeared to have slightly higher levels of zinc within the scallop tissues, with approximately half those levels apparent in the area beyond 150 meters and within the near-field area (immediately downstream of the net-cage system). It is speculated that these elevated levels could be attributable to zinc that has enriched background levels as a result of waterborne dispersion downstream of the farm, but differentially through the upwelling of waste materials at this higher energy site (see Chapter 3).

The near-field region of this site (within 50 meters) was characterized by reduced zinc levels, which may be a result of the infrastructure interference with tidal flows and the displacement of the upwelling effect downstream. However, regardless of the specific processes that have determined the contaminant distribution and environmental partitioning pattern in this study, the effects of oceanographic influences in the dilution and dispersion process is clearly important in defining the potential (and magnitude) for tissue quality impacts within an *integrated*-MTA system.

From a seafood safety perspective, trace metal levels reported across all stations and samples acquired at the study sites (excluding the in-cage stations and reference) were, with few exceptions, within typical tissue concentrations ranges found in commercial seafood. Table 20 compares the mean trace metal levels found at the farm sites, for whole oyster, scallop adductor muscles and scallop viscera, with tissue levels reported in other seafood samples acquired and analyzed by the Canadian Food Inspection Agency (CFIA) between 1999 and 2001.

For whole oysters, trace metal concentrations were within the range of variation reported by the CFIA for oysters from other coastal regions. Levels in oysters are normally higher than in many other seafood product groups, including fishes and other invertebrates (e.g., urchins, shrimp, crabs). Other bivalve species, particularly clams, show higher levels of trace metals such as aluminum and arsenic, but lower levels of the other trace metals (Table 20). Given the lack of significant accumulation in any of these trace metal constituents within their tissues, whole oysters would appear to be good candidates for an *integrated*-MTA system in terms of their negative response to the potential effects associated with trace metal release to the environment from an adjacent finfish aquaculture facility.

Scallops sequestered trace metals, particularly in the visceral tissues, to levels consistently higher than most other shellfish and many of the other seafood groups reported by CFIA (Table 20). Scallop viscera demonstrated substantially higher levels of trace metals than that detected in the adductor muscle, with concentrations of various elements differing between these tissue groups by factors of approximately 2 to 8.

Tissue partitioning of trace metals, which is likely applicable to other contaminant groups, has important implications to product diversity within a shellfish or a potential *i*-MTA system. The bioconcentration of contaminants in specific tissues may exclude specific product types from being processed and offered as seafood alternatives given human consumption risks. Whole scallop, for example, which exceed seafood safety thresholds given very high visceral levels of specific contaminants may be deemed unacceptable for the seafood market, while concentrations in the adductor muscle (meats) may remain well within acceptable levels of risk.

Table 20

Tissue levels of selected trace metals from domestic seafood products in coastal British Columbia. Mean trace metals in scallop and oyster tissue analyzed in this study indicated in red (excluding in-cage and reference). Data From: *Canadian Food Inspection Agency (CFIA) Environmental Contaminants Program (1999-2001).*

			Aluminum (ppm)	Lead (ppm)	Cadmium (ppm)	Zinc (ppm)	Copper (ppm)	Tin (ppm)	Arsenic (ppm)
Bivalve Mollusca	**Butter clams**	*Mean*	21.3062	0.0787	0.1328	14.2737	1.8845	0.2363	2.6167
		95% CI	6.1290	0.0225	0.0283	0.7813	0.1924	0.2331	0.8872
		n	22	22	22	22	22	8	10
	Manila littlenecks	*Mean*	33.8638	0.1213	0.4434	14.2107	1.6176	0.0600	3.0073
		95% CI	7.5312	0.0501	0.0886	0.9902	0.1658	0.0867	0.4856
		n	24	25	25	25	25	16	17
	Oysters	*Mean*	12.9200		2.4700	132.4300	12.0000		1.5700
		n	20		20	20	20		20
	Oysters	*Mean*	14.0782	0.1755	1.9614	181.5071	13.2308	0.2718	1.1575
		95% CI	4.0787	0.1271	0.5289	35.1062	2.1131	0.1876	0.2544
		n	31	31	31	31	31	11	21
	Scallops (viscera)	*Mean*	27.4400		5.5700	31.8000	2.8300		1.0200
		n	20		20	20	20		20
	Scallops (whole)	*Mean*	11.7512	0.0305	2.2542	47.3150	1.3534	0.0010	0.4121
		95% CI	5.9320	0.0089	1.2350	22.4370	0.6416	0.0000	0.1831
		n	14	14	14	14	14	11	11
	Scallops (muscle)	*Mean*	4.6500		1.2900	15.0000	0.3700		0.8500
		n	20		20	20	20		20
	Scallop (muscle)		5.1650	0.0317	0.4225	18.4700	0.4648	0.0010	0.6021
	Geoducks	*Mean*	11.0855	0.0528	0.1939	20.5996	3.2133	0.0010	2.2934
		95% CI	4.6859	0.0249	0.0695	3.1843	0.7407	0.0000	0.8752
		n	26	28	28	28	28	9	20
Other Invertebrates	**Dungeness crab**	*Mean*	3.5380	0.3953	0.3901	34.3088	4.1942	0.3990	11.1806
		95% CI	1.0548	0.7029	0.7032	6.6972	0.9340	0.7322	6.4590
		n	24	25	25	25	25	16	22
	Prawns	*Mean*	0.9004	0.0456	0.0551	11.5994	5.6273	0.0011	7.5277
		95% CI	0.2272	0.0278	0.0612	0.4711	0.9751	0.0001	1.2441
		n	15	15	15	15	15	7	9
	Red Urchins	*Mean*	13.6186	0.0348	0.5449	21.5880	0.4355	0.0010	3.4136
		95% CI	11.7265	0.0359	0.2619	9.9487	0.0491	0.0001	1.0166
		n	5	5	5	5	5	3	5
	Shrimp	*Mean*	4.0044	0.0202	0.0953	8.6748	4.1820	0.0541	3.7106
		95% CI	1.8944	0.0085	0.0237	0.3559	0.7188	0.0649	1.3690
		n	9	9	9	9	9	7	7
Fishes	**Rockfish**	*Mean*	0.7874	0.0099	0.0056	3.3928	0.3662	0.0014	1.2480
		95% CI	0.2845	0.0101	0.0078	0.2641	0.1385	0.0009	0.8287
		n	10	12	12	12	12	9	10
	Salmon, Atlantic	*Mean*	0.3897	0.0061	0.0025	3.6835	0.5707	0.0254	0.2558
		95% CI	0.2580	0.0042	0.0019	0.4215	0.0714	0.0365	0.1250
		n	12	12	12	12	12	11	12
	Salmon, Chinook	*Mean*	0.6074	0.0729	0.0041	3.7414	0.5998	0.0032	0.3787
		95% CI	0.2240	0.0462	0.0027	0.1350	0.1535	0.0027	0.1913
		n	24	25	25	25	25	5	14
	Black cod	*Mean*	0.3456	0.0184	0.0010	2.3423	0.2208	0.0026	0.8122
		95% CI	0.2987	0.0156	0.0000	0.2304	0.1356	0.0013	0.2076
		n	10	10	10	10	10	10	10
	Ling Cod	*Mean*	0.4287	0.0082	0.0041	4.1201	0.3206	0.1353	0.4209
		95% CI	0.4414	0.0121	0.0030	0.4342	0.1134	0.0979	0.3251
		n	7	9	9	9	9	6	6

Despite the clear evidence of partitioning and differential accumulation of trace metals among scallop tissues, this study did not demonstrate a significant effect attributable to contaminant flux from the adjacent finfish operation. The elevated levels of cadmium documented in the visceral tissues, for example, might typically warrant a concern with respect to seafood safety levels (e.g., EU Importation Standard for Shellfish = 1.0 ug/g Cd) but in this Pacific region is the result of normal background levels of such metals. Current government-funded research (Heath, 2004: B.C. Ministry of Agriculture, -Fisheries & Food) is being conducted to examine regional variations in this metal, seasonality, terrigenous inputs, deepwater flux and environmental partitioning.

5.5.2 Shellfish Tissue Quality and Periodic Contaminant Inputs

The potential bioavailability and sea food safety consequences of antibiotic and chemotherapeutic compounds to non-target species in the surrounding marine environment has been identified as one of the major environmental concerns associated with marine netcage culture in coastal British Columbia (BCEAO, 1997). The possible transfer of these antibiotic compounds, and their residues, to wild fish and/or to species of other trophic levels is the basis of concern, with the dispersion and persistence of these compounds considered of importance in the assessment of environmental impacts as well as specific, sea food safety issues. Knowledge of the fate and effects of these potential waterborne contaminants is also essential in evaluating the technical feasibility of integrated aquaculture.

The application of antibiotic and/or chemotherapeutic compounds, in the context of prescribed fish treatments, represents a potential periodic contaminant flux at a finfish aquaculture facility that would also extend to the operation of a proposed *i*-MTA system. Given the short-term delivery of these compounds (typically $\leq$10 days), and the low concentrations of the active ingredient included within the feed, the water column bioavailability and localized persistence of these contaminants were expected to be limited at the onset of this research initiative.

Shellfish tissue accumulation of the antibiotic oxytetracycline (OTC), applied over a number of treatments at one of the study sites (Young Passage), demonstrated a measurable downstream effect that comprised both a temporal and a spatial component. In terms of spatial effects, the detection of OTC in shellfish tissues (oysters and scallops) was evident to a distance of approximately 150 meters downstream of the finfish aquaculture facility. Tissue concentrations declined over this distance, suggesting a dilution process that progressively limited uptake for these animals along the survey transect.

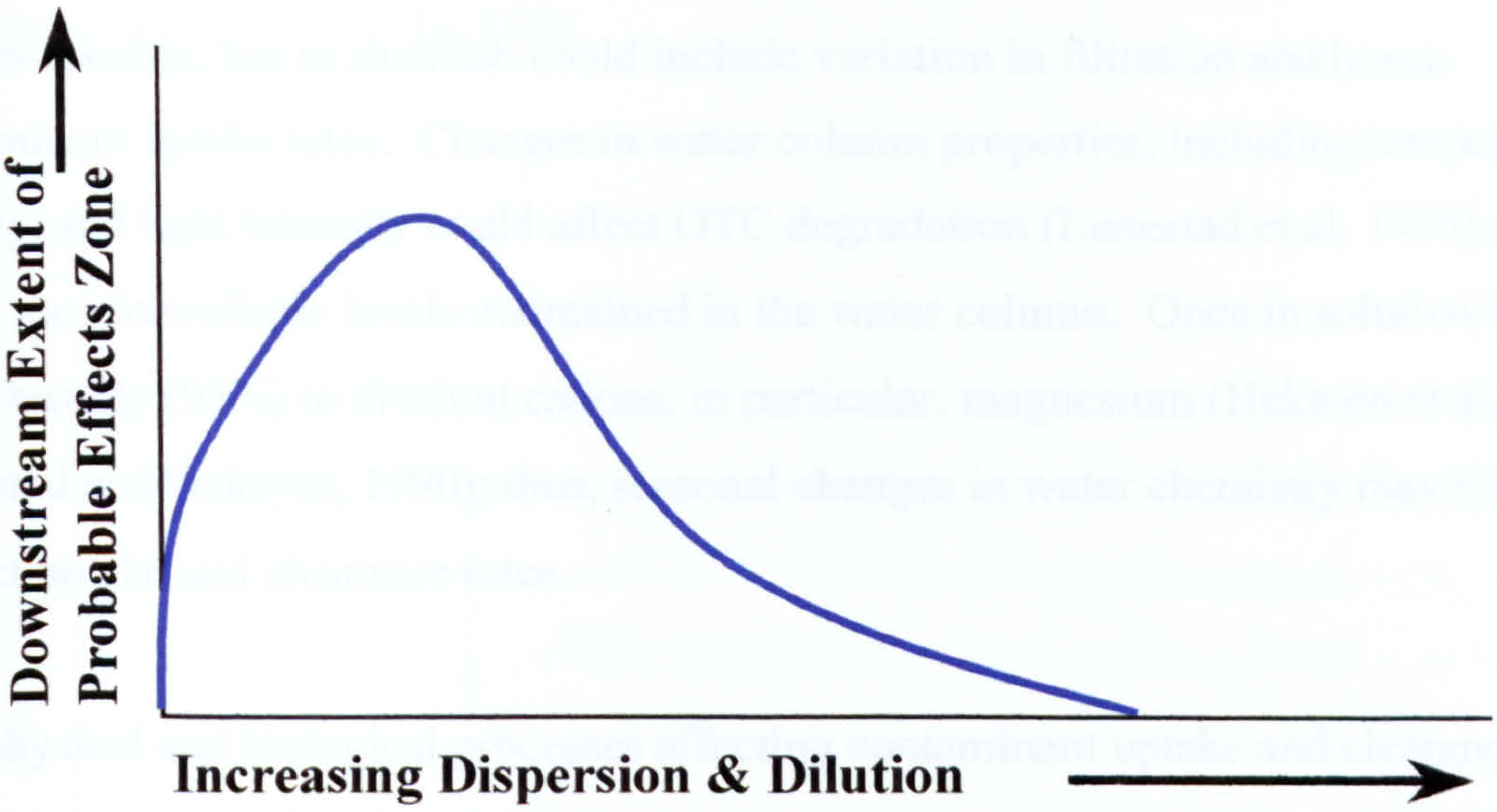
A distinct zone of influence for these waterborne contaminants, determined by the site-specific oceanographic characteristics defining the dilution and dispersion process, will dictate the extent of a *Probable Effects Zone* for a farm site (Figure 61). It is assumed that as tidal energy increases so does the dispersion and dilution of any waterborne contaminants that might be released from the finfish system. At some distance downstream, the *in situ* concentrations would become sufficiently small that bioaccumulation in the shellfish (the measured effect) would no longer be detectable, or presumably of concern. Further increase in the dispersion/dilution energies would result in a decreased *probable effect zone* that would, theoretically, approach a 'no effect' threshold when all of the waterborne contamination is dissipated at a rate that exceed the uptake capability of the shellfish species. This process is, of course, also dependent on the magnitude and duration of the contaminant flux.

In the context of an *i*-MTA system, the *Probable Effects Zone* would represent the area within which a water quality and a shellfish tissue management program would need to be applied in order to ensure that product safety (tissue) standards are met prior to harvest. As such the relationship with oceanographic characteristics plays an important role in site selection for *i*-MTA, as well as in defining infrastructure configuration and in establishing management program associated with short-term harvest restrictions due to potential contaminant pulses.

The temporal aspect of OTC residue effects on adjacent shellfish appeared evident in both uptake as well as clearance rates for each of the two test species. In both oysters and scallops the uptake of OTC was detected shortly after treated feed was delivered to the fish, supporting previous studies that have suggested a large proportion of the antibiotic is passed directly through the fish, unutilized (Samuelsen et al, 1992).

Figure 61:

Extent of *Probable Effects Zone* as a function of site-specific oceanographic characteristics (dispersion/dilution).



The present study estimated that OTC loss to the water column was over 98%, although the associated release rate over the treatment and post-treatment periods was not predicted (Chapter 4). However, for the scallop samples an estimate of the OTC uptake and clearance rates were established for two OTC treatments. During a 10-day winter treatment, an OTC uptake rate of 0.056 ug/g/day was sustained until peak shellfish tissue concentrations were achieved within a week following the treatment. Clearance of the OTC from the shellfish occurred at a rate of 0.016 ug/g/day, extending the effect to a total of 32 days.

Analysis of a second OTC treatment, completed during the late summer, revealed a response significantly different in terms of residue uptake and clearance in the scallops. In this case, uptake was estimated at 0.100 ug/g/day while clearance occurred at a rate of 0.109 ug/g/day. Despite an OTC application of approximately twice that of the first treatment, the resulting tissue concentrations achieved in the shellfish was similar, or slightly less, along the entire downstream transect.

The comparison of uptake-clearance dynamics suggests a significant seasonal component to these processes. While the summer uptake rate was approximately twice that of the winter rate, the OTC clearance rate in the summer was almost 10-fold faster.

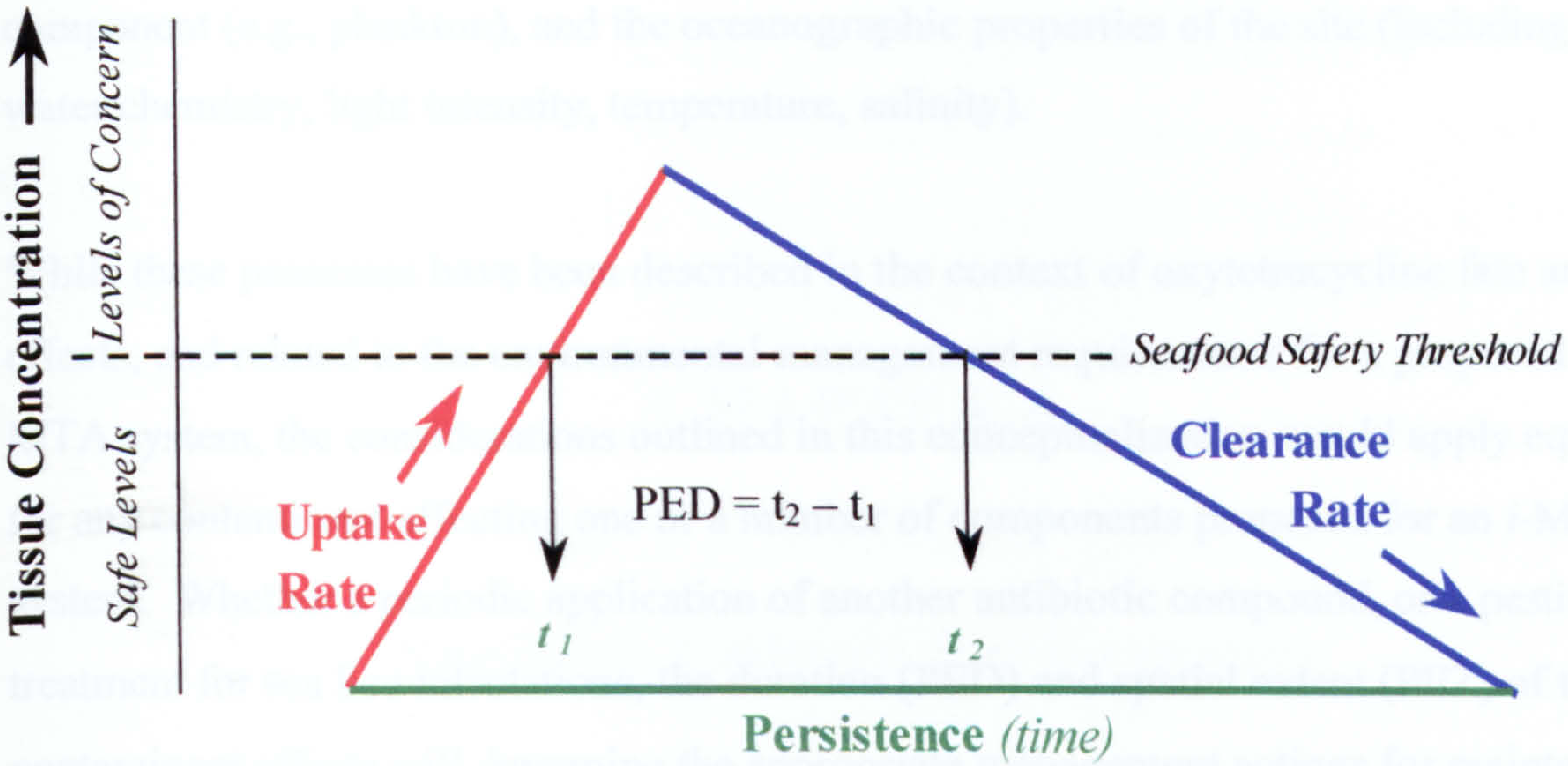
Differences in physiological activity between summer and winter months will be species-specific, but in shellfish could include variation in filtration and hence contaminant uptake rates. Changes in water column properties, including temperature, salinity, and light intensity could affect OTC degradation (Lunestad et al, 1995) and hence the bioavailable levels maintained in the water column. Once in solution OTC binds heavily (95%) to divalent cations, in particular, magnesium (Hektoen et al, 1995; Lunestad and Goksyor, 1990); thus, seasonal changes in water chemistry may also impact uptake and clearance rates.

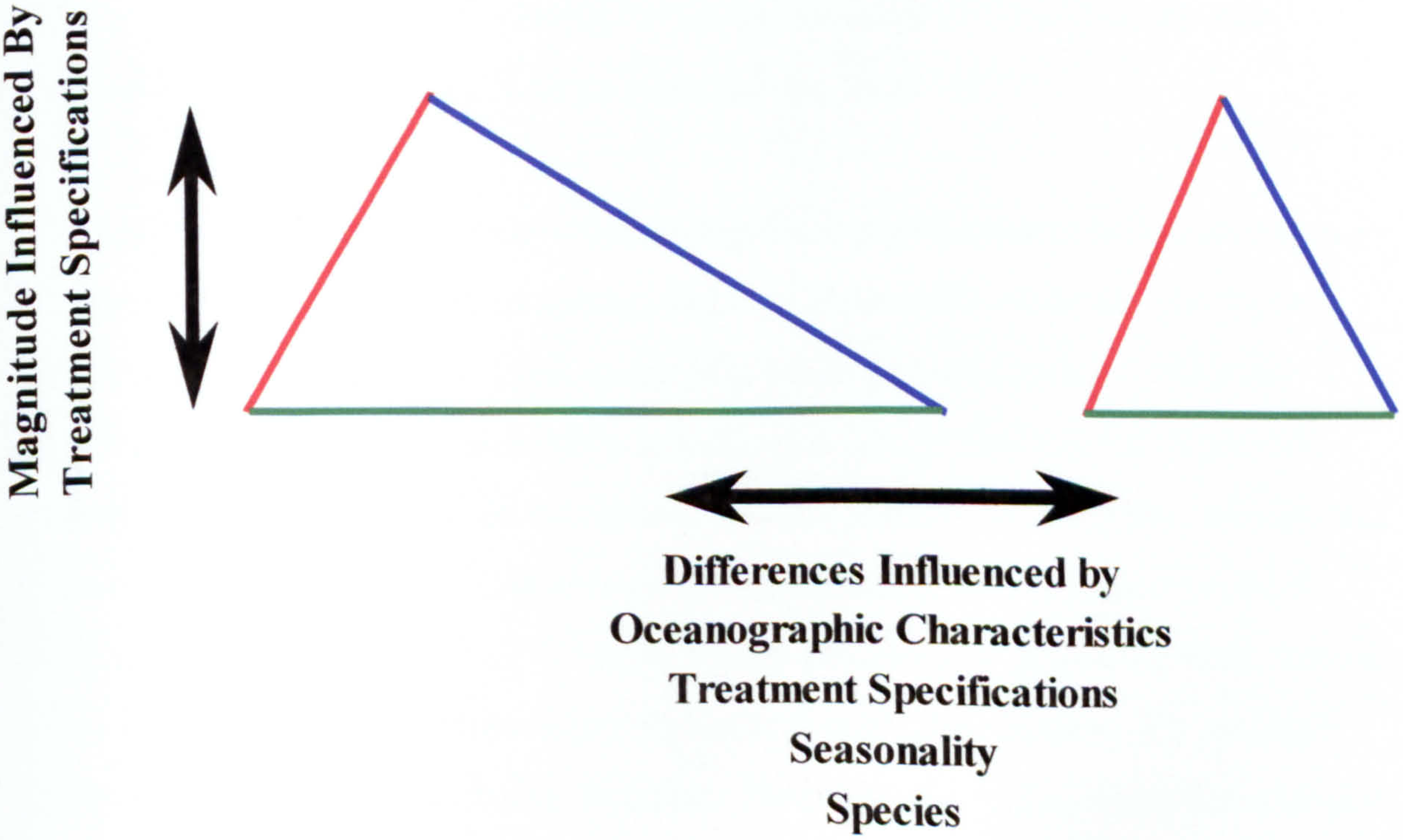
The physical and biological processes affecting contaminant uptake and clearance rates are important considerations in the management of a proposed *i*-MTA system. Where traditional monoculture systems would not typically require consideration of external contaminant interactions, the development of an *i*-MTA system must recognize the periodic impacts such inputs.

Figure 62 provides a conceptualized diagram that illustrates the relationship among the variables that would define the *Probable Effects Duration* (PED) for such considerations. The process variables include concentration (contaminant flux; bioavailable or *in situ* levels), tissue uptake rate, tissue clearance rate, and time (contaminant persistence). Slow uptake and/or clearance rates will result in prolonged, or persistent tissue levels of a contaminant. The proportion of time during which tissue concentrations exceed an established seafood safety threshold ($t_2 - t_1$) has been termed the *Probable Effects Duration* (PED), and represents the time during which tissue concentrations would warrant management action in an *i*-MTA operation (e.g., cessation of scheduled harvests during PED; relaying of product for depuration).

Figure 62:

Generalized contaminant uptake and clearance diagram illustrating the relationship with environmental persistence, concentration and Probable Effects Duration (PED). Lower diagram provides a summary of the factors that will influence uptake and clearance rates, as well as the PED.





The PED will vary in response to a number of additional variables. As shown in the lower portion of Figure 62, the shape of the Uptake-Clearance-Persistence plot will be influenced vertically by changes in the contaminant input (treatment specifications within the finfish component of an *i*-MTA), and horizontally as a result of parameters such as species under culture, physiological status, age, seasonality, a competing biotic component (e.g., plankton), and the oceanographic properties of the site (including water chemistry, light intensity, temperature, salinity).

While these processes have been described in the context of oxytetracycline fate and effects, and related to the environmental management requirements for a proposed *i*-MTA system, the considerations outlined in this conceptualization would apply equally for any contaminant affecting one or a number of components proposed for an *i*-MTA system. Whether a periodic application of another antibiotic compound, or a pesticide treatment for sea lice infestations, the duration (PED) and spatial extent (PEZ) of the contaminant effects will determine the appropriate management actions for maintaining product quality and ensuring compliance with seafood safety standards.

CHAPTER 6

Shellfish Culture Performance

6.1 Introduction

In 1997, the British Columbia Salmon Aquaculture Review (EAO, 1997) completed an exhaustive evaluation of the western Canada salmon aquaculture industry that, in addition to summarizing the existing state of scientific evidence regarding environmental effects of this coastal industry, included the consolidation of public opinion (substantiated concerns as well as perceptions) associated with the environmental performance of this relatively new coastal industry sector. Included within the latter component of this Review were the concerns of coastal First Nation peoples, which were eventually summarized within a separate volume of the final Review document (EAO, Vol. 2).

These social concerns represent an important factor in the development and ongoing operation of salmon farming in coastal British Columbia. Yet these environmental issues, summarized in the Aquaculture Review (EAO, 1997; Vol. 2), have been very difficult to address, particularly in light of the "*mistrust of scientific evidence collected to date and the conclusions drawn*" (First Nations perspectives, EAO, 1997). These concerns have been further articulated in statements suggesting "*skepticism regarding existing information was due to limited or no involvement of First Nations with research or the results*" (First Nations perspectives, EAO, 1997).

Although the scientific approach to addressing these environmental questions remains an appropriate framework for acquiring objective, defensible evidence, the direct involvement of societal parties that continue to focus on perception becomes an important and necessary consideration in a research design that intends to provide evidence that will ultimately be recognized by such parties. In this study, the detailed examination of waterborne contaminants to adjacent shellfish resources involved complex scientific methodology (Chapter 5) that precluded such involvement, but did result in a series of conclusions regarding seafood safety implications for shellfish grown adjacent to salmon farming facilities. However, given the entrenched attitudes and opinions regarding the environmental impacts on shellfish, it remains unlikely that these results, alone, would ever be sufficient to alleviate such social opinion.

Despite demonstrated safety of shellfish grown adjacent to salmon farms, in terms of quantifiable levels of potential contaminants (e.g., trace metals, antibiotic residues), the argument that these resources remain affected and are "different" as a result of these finfish operations would continue unaffected by such scientific evidence. The belief that large volumes of farm-derived organic material are impacting shellfish resources would most likely be argued, if not from chemical residues, then from a taste, odour, and palatability perspective.

The evaluation of potential taste and odour problems can be effectively addressed using organoleptic testing techniques developed by the food industry. These tests typically comprise a variety of approaches (e.g., blind, double-blind) to ensure an unbiased assessment and analysis of results. Robinson *et.al.* (*in prep.*) conducted an organoleptic evaluation of mussels (*Mytilus edulis*) grown along the side of salmon netcages in the Bay of Fundy (eastern Canada). This study, completed with some 20 volunteers, demonstrated that participants could not discern which samples came from the salmon farm and which were acquired from a geographically removed culture facility.

The use of organoleptic tests can provide a useful method by which social issues, such as those of the British Columbia coastal First Nations, can be addressed through direct participation. Organoleptic tests are very simple to design, to implement, and to understand in terms of the results. The simple nature of such tests, and the potential for considerable direct involvement, makes the outcome difficult to refute.

If one assumed that the '*tainting*' and quantifiable water quality interactions between salmon farm facilities and other integrated aquaculture components (such as shellfish) were minimal, if not nonexistent, then the potential for integrated multi-trophic aquaculture could be assessed simply through consideration of the culture performance of each of the components. In the case of integrated finfish-shellfish aquaculture, the growth and survival of the selected species would become the important consideration from a commercial development perspective.

Chopin et.al. (1999) and Neori et al. (2004) both reported significant increases in macrophyte (*Laminaria saccharina*) and shellfish (*Mytilus edulis*) grown in close proximity to a salmon aquaculture facility. The production of the macrophyte at the farm site was 46% higher than at a station 1250 meters away. The mussels grew 100% faster at the stations located immediately adjacent to the salmon net-cages. Given that the authors also found no accumulation of antibiotic residues or phytotoxins within these integrated culture components, the case for considering multi-trophic aquaculture systems was supported.

The interest in developing Multi-Trophic Aquaculture (MTA) will, given the commercial nature of such a venture, be encouraged with trial results that demonstrates the production potential shown by Chopin et.al. (1999). The environmental benefits associated with a balanced MTA system, in reducing net waste flux to the receiving environment, is also worthy of consideration. In the MTA system studied by Chopin et.al. (1999), the fed aquaculture component of the system, (salmon) was supplemented by an extractive inorganic component (macrophytes) and an extractive organic component (mussels). Requiring feed for only one of the three production components, the system capitalizes on the wastes from the first to support production of the other two.

6.2 Objectives

In an effort to address the question of shellfish culture performance at the selected salmon aquaculture facilities, a routine assessment of the shellfish deployed at each farm site station was completed. The specific objectives of this study component were to determine:

- whether shellfish growth was significantly different as a result of proximity to the finfish aquaculture systems;
- the spatial and/or temporal extent of shellfish growth effects (if they were detected);
- if proximity to the finfish operation affected survival of the shellfish species;
- the spatial and/or temporal extent of shellfish lethality effects (if they were detected);

- if the shellfish tissues were tainted as a result of exposure to farm-derived organic materials; and to determine
- how these growth, survival and organoleptic data compared with shellfish performance data acquired from reference growing areas.

The implications of the shellfish culture performance data are subsequently discussed in terms of integrated finfish-shellfish aquaculture.

6.3 Methods and Materials

The evaluation of shellfish culture performance was completed using a combination of direct measurements of growth and survival, as well as through an independent organoleptic study to examine the potential for tissue tainting. The specific methods employed for these study components are described below.

6.3.1 Shellfish Growth and Survival Measurements

Shellfish growth and survival were measured at each of the two study farm sites at an approximately 45-day interval following deployment of seed. Shellfish seed was acquired from two local shellfish growers, scallops from one and oysters from another, ensuring that the stock were from the same year-class and from a location at least 10 km from any salmon farm sites. The survey schedule corresponded to sampling of the shellstock for analytical purposes, and resulted in 8-10 surveys over the duration of the study.

The growth for the scallop (*Patinopecten yessoensis*) and oyster (*Crassostrea gigas*) comprised a shell height measurement conducted over the long axis of these animals. At each downstream station a total of ten animals were randomly selected from the suspension infrastructure (nets, trays), measured (shell height in mm), and returned to their respective positions. Data recorded in the field included the measure of shell height, as well as observations related to growth (e.g., shell deformities, excessive fouling, etc.).

In terms of mortality, the cages accessed randomly for the shell growth measurements were examined, in their entirety, for the presence of dead animals. Each of these animals was measured for shell height, and these data post-analysed to assess the occurrence and frequency of mortality over the growth period. It was assumed that death of the animals could be related to the specific time within this period by identifying the corresponding portion along the growth curve (by size) at which these mortality observations fell. This approach avoided a requirement to examine all shellfish nets and cages, during the individual survey periods, and thus limited the stress imposed on these animals.

Growth and survival data were compiled in a common database and examined for spatial and temporal trends in shellfish culture performance that could be attributed to proximity with respect to the salmon farms. Data were standardized as growth rates (shell height increase in mm/month) and presented as growth curves over the length of the survey period to differentiate seasonal change from spatial changes in this performance metric. Mortality data were related directly to growth curves (shell height in mm) using data averaged across each of the two study sites.

6.3.2 Organoleptic Evaluation

The organoleptic evaluation was completed at the end of the field program for this study. The timing for this component of the culture performance evaluation thus ensured that the animals used for the "taste-test" had been exposed to any potential tainting agent(s), originating from the adjacent salmon aquaculture system, for the longest possible period. Although the original intent was to include both oysters and scallops in this assessment, the numbers of remaining oysters fell short of that required for the test. Consequently, only scallops were used for this organoleptic evaluation, leaving the tests of tainting on whole-animal consumption incomplete.

Despite this missing component to the test, a parallel polyculture research program in eastern Canada was using mussels as whole animal indicators for assessing integrated aquaculture potential, with this species also being used to evaluate the possible effects of tainting. The results from this program (Robinson et.al., in prep.) did provide an opportunity to supplement the west-coast information with data related to whole-animal consumption evaluation.

To address the concerns that First Nation peoples have expressed over salmon farming, and the environmental impacts they perceive are occurring on their traditional shellfish fishery resources, the organoleptic component of this study was conducted entirely with representatives of these coastal peoples. The study was conducted at the remote village of the K̲a:'yu:'k't'h' / Che:k̲:tles7et'h̲' First Nation, located on the northwest side of Vancouver Island, and employed 22 community volunteers for the organoleptic testing procedure.

To ensure that data remained unbiased, First Nation representatives participated in all aspects of the test, including the preparation of shellfish samples, the taste testing itself, and the data compilation and validation. The taste-testers remained separate from all other preparatory activities, maintaining a 'blind' aspect to the procedure.

Samples of scallop meats (adductor muscle) were poached briefly in water (60 seconds), avoiding use of any spices that could mask the inherent flavor and texture of the portions used for the test. Size of these samples was also standardized, maintaining easily managed portions of approximately 1 x 1 x 1 cm. The scallop samples were acquired from the Young Passage farm site, given that this site had shown some degree of tissue accumulation of farm-derived materials (impact), as well as from a local control area (Baynes Sound; 50 km south of Young Passage). The control scallops were also of the same year-class as the study site animals, eliminating any potential effects related to age, exposure time, and/or regional life history experience.

The design of the blind organoleptic test itself was very basic, allowing a simple evaluation of scallop samples by each taste tester. A small, partitioned plate contained three scallop portions, allocated in one of four possible scenarios, shown in Table 21. Each plate was accompanied by a data sheet that allowed the tester to simply identify the distribution of samples as shown in Table 21. The record sheet also included written instructions for the tester (if unclear in the verbal presentation provided), as well as a Test Number (unique to that test scenario) that allowed assignment of the test results to the database.

Table 21: Shellfish portion delivery scenarios, showing position of study site (Young Passage) scallop (1) with respect to Control portions (0). The fourth scenario shows all 3 portions from the Control area.

Position1	Position 2	Position 3
0	0	1
0	1	0
1	0	0
0	0	0

The taste testers were asked to evaluate each of the three scallop portions to determine which one, if any, was different from the others. The volunteers were asked to consider taste, in terms of bitterness or sweetness, smell, as well as the texture of the three portions (e.g., stringy, tough, soft, etc.).

A total of 22 taste testers volunteered for the study, with each requested to complete three taste test scenarios. The individual tests were assigned randomly by a volunteer who was not involved with sample preparation (cooking, portion allocation), or with data compilation aspects of the test. This volunteer also acquired the completed data sheets and relayed them to the researcher for compilation.

Upon completion of the taste tests the recorded information were entered into the developed database, ensuring that test results were compiled within the record corresponding to the correct Test Number. To validate these data, and to ensure that no errors had been made in this transcription, another First Nation volunteer was retained to randomly select raw data sheets and to compare the recorded information with that complied within the database for subsequent analysis.

Given the limited sample size (n=3) for individual taste testers, any possible within-tester effects were not statistically assessed. All data were treated independently, and it was assumed that these data provided a reasonable population estimate (in a statistical sense) of the organoleptic response for these First Nation participants. The reported responses of the taste testers were statistically compared with those of the actual portion presentations to determine whether volunteer responses could be accounted for by a process other than chance. The data were also summarized as percentage response for discussion purposes.

6.4 Results

The culture performance of the scallop (*Patinopecten yessoensis*) and the Pacific oyster (*Crassostrean gigas*) was evaluated through growth and survival data collected during a monitoring period that extended from September/2001 through August/2003. The organoleptic component was completed at the end of this monitoring period, in October/2003, to provide animals for the test that had experienced a long exposure time to any near-field contaminant loads, and thus provided a worse-case scenario of the possible water quality impacts in a potential integrated aquaculture system. The following sections summarize the results of these observations for each species, independently.

6.4.1 *Patinopecten yessoensis* Growth and Survival

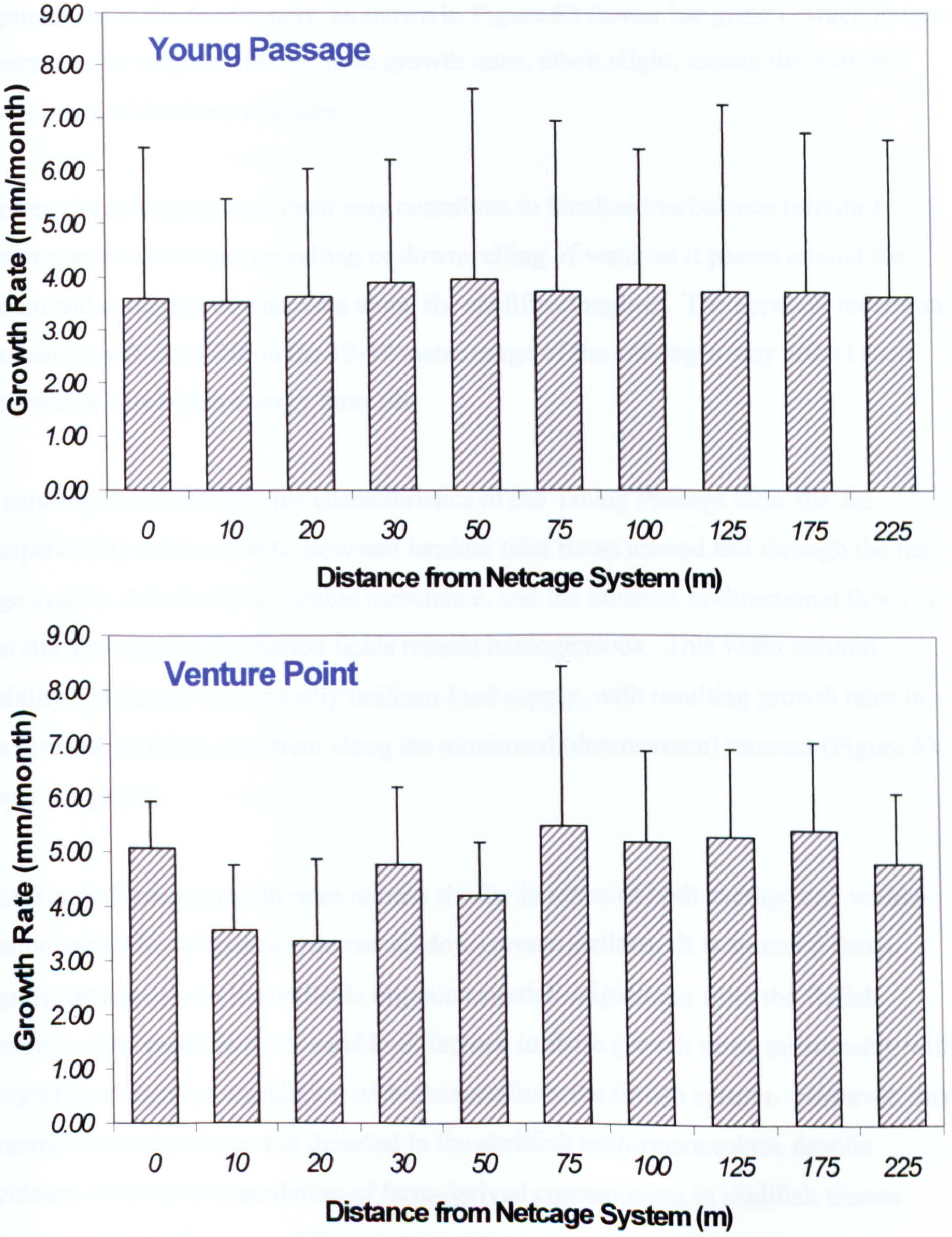
The Japanese scallops were deployed as 17.2 mm seed (mean; n=50) at the Young Passage farm site in September/2001 and as 36.2 mm seed (mean; n=50) to Venture Point in May/2002. The entry delay at the latter site was due to the fallow status of the farm site in late 2001. A new production cycle was initiated at Venture Point in January/2002, but scallop seed were not available until end of April/2002.

Figure 63 compares the growth rate of the scallops deployed at each of the study sites, and at each sampling station downstream of the salmon farm system, expressed as mm shell height per month. These estimates show the growth rate, calculated as means of the site survey periods, with growth rate variability as the sample standard error.

Although a one-way ANOVA suggests that there was no significant difference among the growth rates for the ten stations positioned downstream of either of the two study sites ($P<0.05$), the mean growth rates (ignoring within-station variation) did appear to fluctuate more among the Venture Point site stations than at those assessed at the

Figure 63:

Patinopecten yessoensis growth rate determined using data acquired over entire production cycle at the Young Passage (upper bar graph) and Venture Point (lower bar graph) study sites. Values for each station represent sample means with associated standard error estimates.



Young Passage farm. It is speculated that this slight difference in performance could be attributable to the different hydrodynamic characteristics of the sites.

At the Venture Point, the higher current velocities and more complex bathymetric features (see Chapter 3) will produce a more turbulent water column, resulting in a mixing of available food resources (seston) to the shellfish. This turbulence may lead to patchiness in the food supply, as shown in Figure 63 (lower bar graph), which in turn is expressed as a spatial difference in growth rates, albeit slight, among the stations downstream of the farm structure.

Further, the netcage system itself may contribute to localized turbulence (mixing) patterns, and cause some upwelling or downwelling of water as it passes around the system and continues downstream along the shellfish longline. The apparent reduction in mean growth rates within the 10-30 meter range of the net-cages may reflect such processes at this higher energy farm site.

In contrast, the oceanographic characteristics of the Young Passage farm site are comparatively subdued, with slow and laminar tidal flows around and through the net-cage system. A lack of measurable turbulence, and the inherent bi-directional flows at this site also ensures that seston fields remain homogeneous. This water column stability is reflected in a spatially uniform food supply, with resulting growth rates in the resident shellfish consistent along the monitored (downstream) transect (Figure 63, upper bar graph).

The Young Passage growth rates remain similar in terms of both average and within-station variability (Figure 63) across all downstream stations. It is assumed that a significant increase in bioavailable organic material, originating from the finfish system, would result in a measurable difference in these growth rates, presumably with a significant negative correlation with distance from the finfish system. However, this organic waste signal was not detected in the shellfish performance data, despite evidence of the bioaccumulation of farm-derived contaminants in shellfish tissues situated within 150 meters of the net-cage system (see Chapter 5).

Evaluation of the shellfish culture performance data are examined further using contour plots of shellfish growth rate as a function of distance from the finfish net-cage system (spatial pattern) and sampling period (temporal change). Figures 64 and 65 present color contour plots of shellfish growth rate data for the Venture Point and Young Passage farm sites, respectively. Although the temporal and spatial scales are maintained for each plot, for comparative purposes, the color scale for growth rates are different and reflect the significantly higher growth rates observed at the Venture Point farm site.

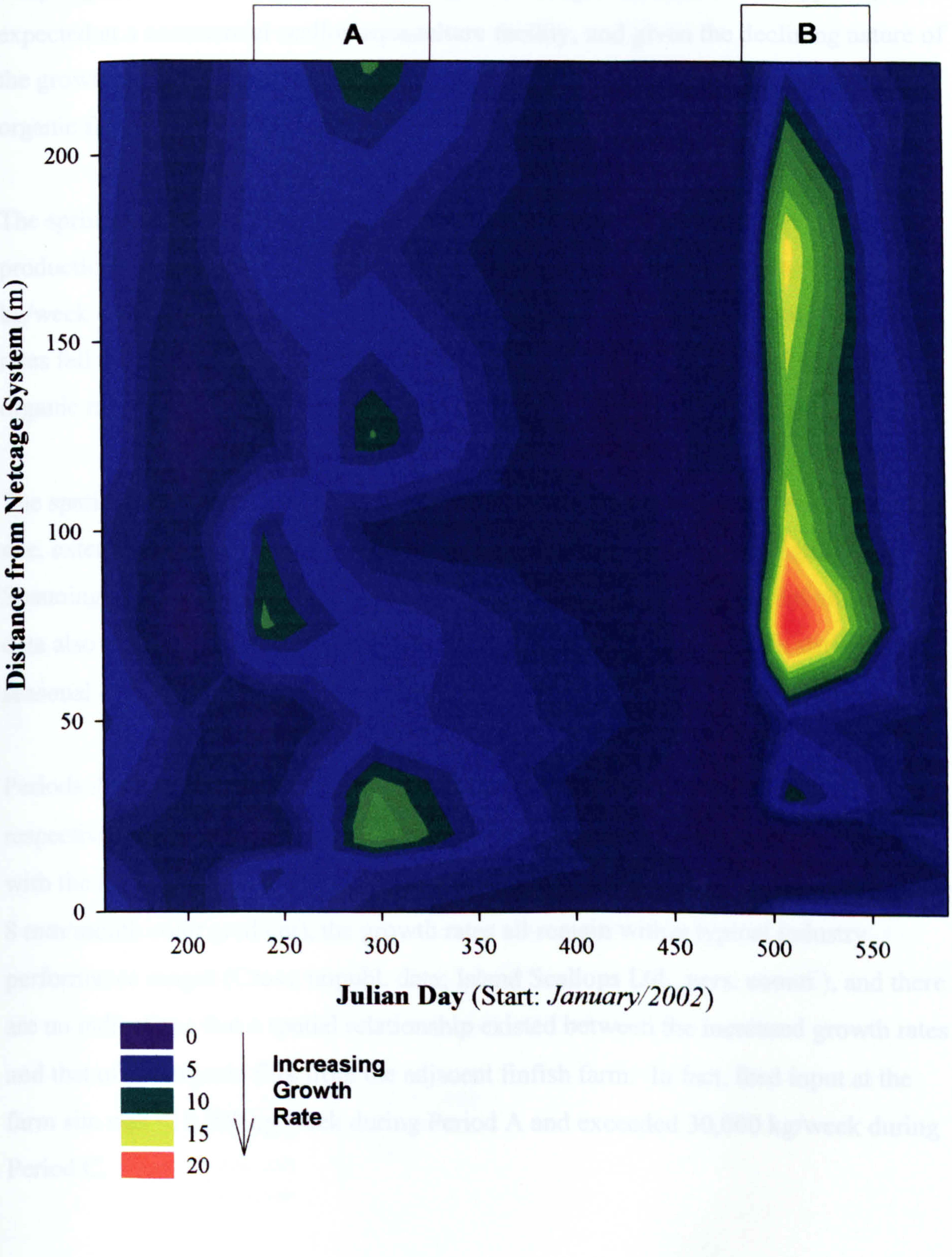
In Figure 64, Venture Point growth performance for the deployed scallops are shown for the period January/2002 through August/2003. The growth contours (mm/month), for this study site, were extrapolated from data acquired from the 8 survey periods conducted across the 10 sampling stations established along the 225-meter downstream transect, i.e., a total of 80 growth rate measurements, each of which was based on the average growth of ten animals.

Increased growth rates are revealed for two periods over this time frame; the first (A, in Figure 64) in the late summer (August-October, 2002) and the second (B, in Figure 64) in the spring April-May, 2003). These periods correspond to natural, seasonally occurring elevations in phytoplankton with the first represented by a late summer bloom, typically of flagellated forms (e.g., *Heterosigma* sp.), and the April-May (or what is commonly referred to as the 'spring bloom') comprised of diatomaceous species (e.g., *Chaetocerus*).

The late summer bloom at Venture Point (Period A) indicated that although scallops responded to this increase in available phytoplankton with a growth rate of between 5 and 10 mm/month, opposed to the winter rates of <5mm/month, there was no clear indication that a spatial relationship existed between this increased growth and the adjacent net-cage system. This growth remains within that expected of scallop performance at commercial aquaculture operations (Island Scallops Ltd., pers.comm.), and does not suggest any enhancement effects from the adjacent finfish operation.

Figure 64

Contour plot of *Patinopecten yessoensis* growth performance at Venture Point study site. Growth data standardized as change in average shell height (mm/month; n=10) across each of the 8 survey periods (X-axis = Julian Day starting at January 1, 2002) for each of the 10 longline stations (Y-axis). **A**: August – October, 2002. **B**: April-May, 2003.



The spring bloom that occurred in April-May/2003 (Period B, Figure 64) appeared to have a more pronounced effect on scallop growth at the Venture Point farm site than that of the previous summer event. Despite minimal growth immediately adjacent to the net-cage system (within 50 meters), growth rates exceeded 10 mm/month downstream of the farm, and actually peaked at 15-20 mm/month at the 75-meter sampling station. These extreme rates of growth are greater than what is typically expected at a commercial scallop aquaculture facility, and given the declining nature of the growth pattern downstream along the survey transect the possible effects of the organic flux from the farm is brought into question.

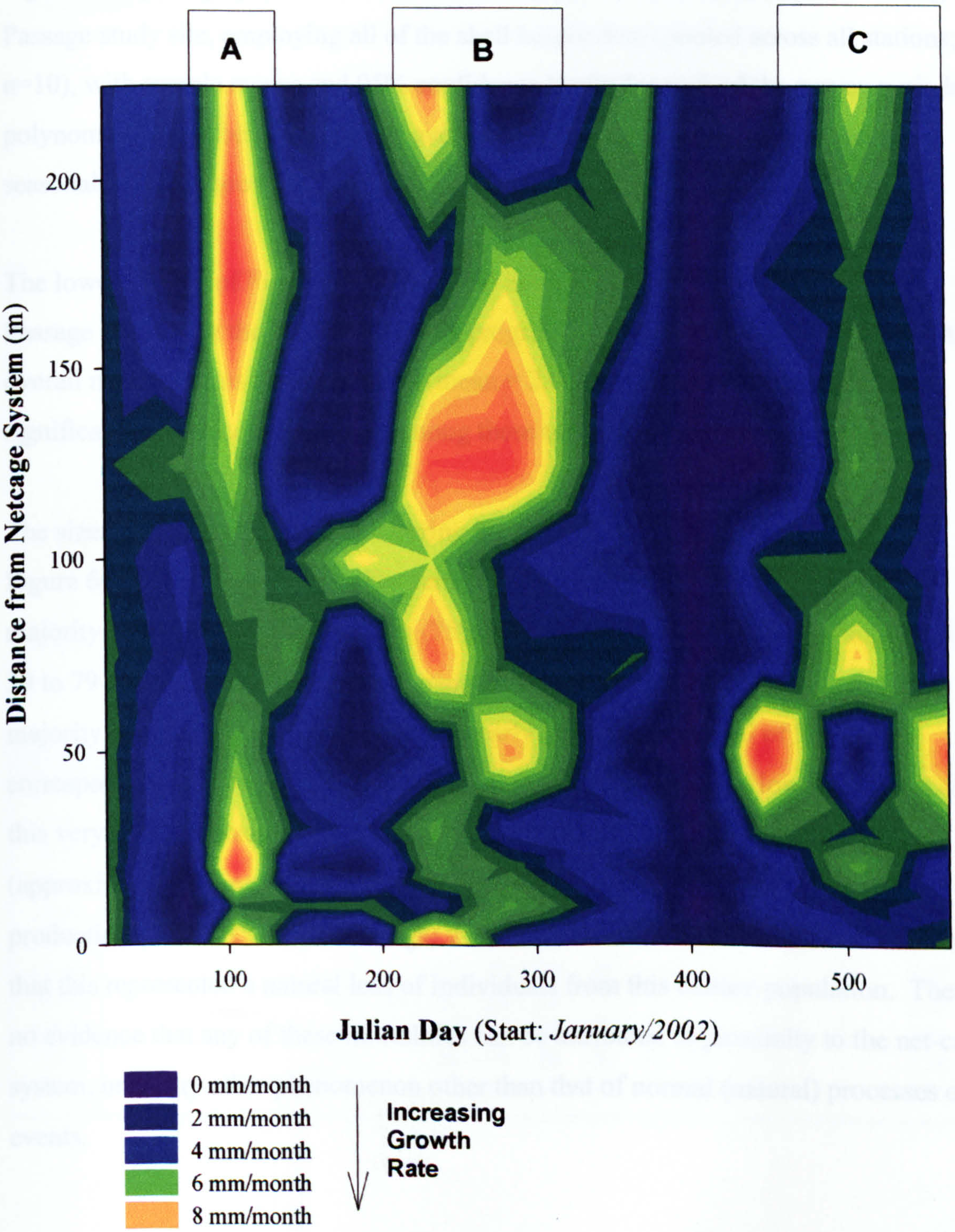
The spring growth at Venture Point also occurred prior to and at the onset of peak production at the finfish farm, with feed inputs increasing from 45,000 to 65,000 kg/week over this period. However, given that the peak feed input occurred as growth rates fell back to levels 5-10 mm/month, it is unlikely that the anticipated flux of organic material from the farm was responsible for these observed growth rates.

The spatial and temporal changes in scallop growth rates for the Young Passage farm site, extending from January/2002 through August/2003, are shown in Figure 65. Spanning a longer survey period than that conducted for the Venture Point site, these data also demonstrate the fluctuations in scallop growth rates that can be attributed to seasonal changes in phytoplankton populations.

Periods A and C correspond to the spring blooms that occurred in 2002 and 2003, respectively, while Period B illustrates the increased growth rates that are associated with the late summer blooms of August-October, 2002. Provided at a reduced scale (0-8 mm/month color gradient), the growth rates all remain within typical industry performance ranges (Cross, unpubl. data; Island Scallops Ltd., pers. comm.), and there are no indications that a spatial relationship existed between the increased growth rates and that of the organic flux from the adjacent finfish farm. In fact, feed input at the farm site was <10,000 kg/week during Period A and exceeded 30,000 kg/week during Period C.

Figure 65:

Contour plot of *Patinopecten yessoensis* growth performance at Young Passage study site. Growth data standardized as change in average shell height (n=10; mm/month) across each of the 12 survey periods (X-axis; Julian Day starting at January/2002)) for each of the 10 longline stations (Y-axis). **A**: April-May, 2002. **B**: August-October, 2002. **C:** April-May, 2003.



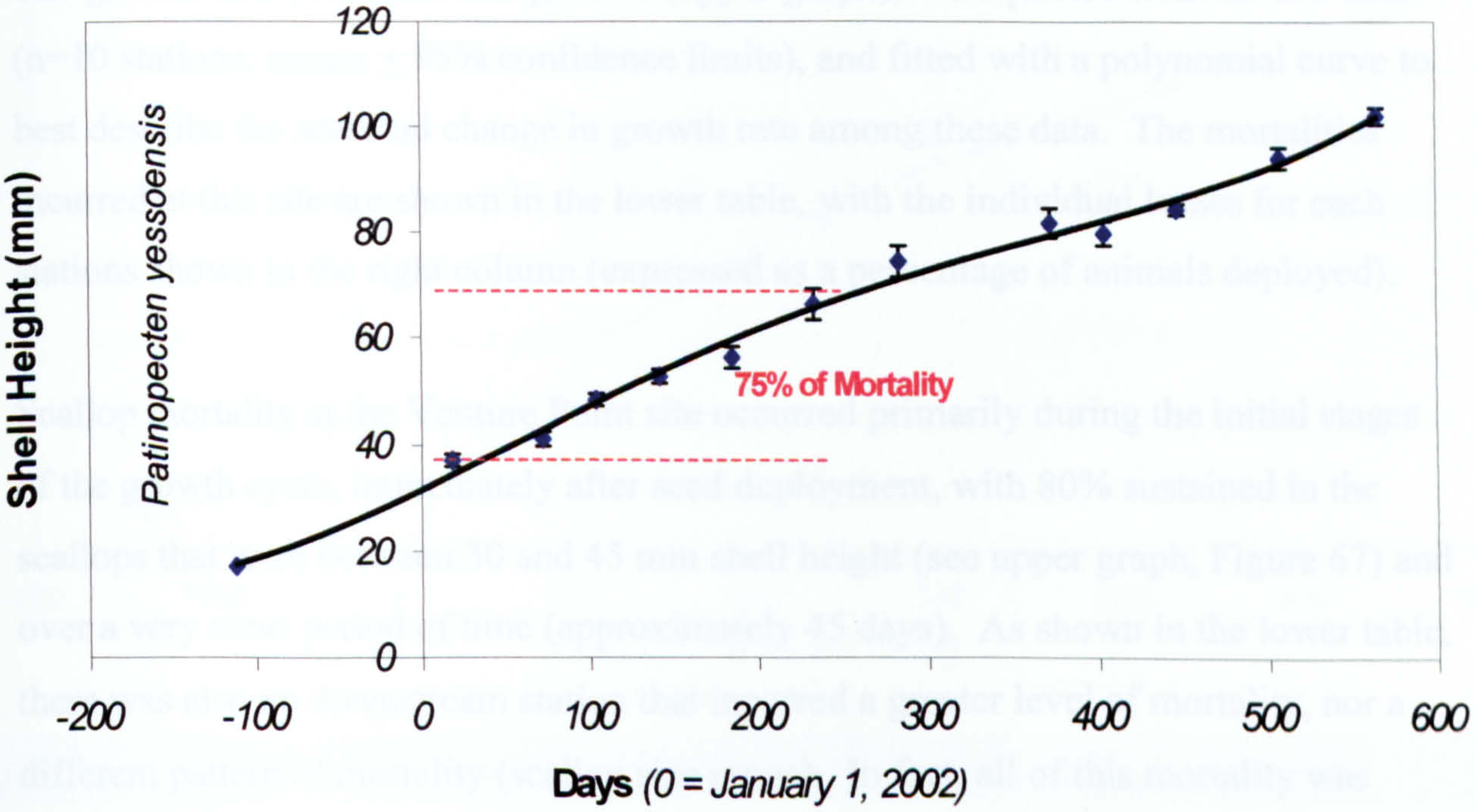
The scallop mortality observed at each of the study sites was also comparable to that experienced in the British Columbia shellfish industry (Island Scallops Ltd., pers.comm.) with estimated losses of less than 10% over the growout period (seed to full-size adult).

Figure 66 (upper graph) summarizes the scallop growth information for the Young Passage study site, employing all of the shell height data (pooled across all stations; n=10), with sample means and 95% confidence limits for each of the survey periods. A polynomial line of best fit is placed over these data to approximate the observed seasonality in this growth.

The lower portion of Figure 66 presents the mortality data acquired at the Young Passage site during the routine sampling program. For each of the downstream stations overall mortality was typically < 3% (mean = 2.08; SD = 0.56), and there was no significant difference in mortality among these ten stations (P<0.01).

The sizes of the animals that died during the study are shown in the lower table of Figure 66, along with the mortality summary statistics (%) discussed above. The majority of these scallop mortalities (75%) occurred within a size group ranging from 39 to 79 mm shell height. In transferring this range to the upper growth curve, the majority of mortality took place between early spring and late fall of 2002, corresponding to a period of increased growth in these scallops. However, given that this very low percentage of growout mortality occurred over such a long period (approximately 8 months), and that it happened during a period of relatively low production (in terms of feed loadings) within the finfish component, to it is assumed that this represented a natural loss of individuals from this culture population. There is no evidence that any of these mortalities can be attributed to proximity to the net-cage system, or to any other phenomenon other than that of normal (natural) processes or events.

Figure 66: Growth of Japanese scallop (*Patinopectin yessoensis*) at Young Passage farm site showing incurred mortality in test animals (lower table) and the period during the production cycle in which mortality occurred (upper graph, section identified in red). Growth curve shows measured shell heights (n=10) with 95% confidence limits; polynomial fit illustrates seasonal growth rate increases.



	Individual Sizes for Each Recorded Mortality (mm)														% Mortality
0-m					26	37	54	65	68						**2.00**
10-m						44	55	62	63	65			74		**2.40**
20-m					28	36	60	68	72						**2.00**
30-m								68	73	73			76		**1.60**
50-m	5	7	10	16				66	66					80	**2.80**
75-m							57	65	68	68					**1.60**
100-m					24	31	62	64	68						**2.00**
125-m							60		69	70	78				**1.60**
175-m					25		58	63	68	69	70	72	75		**3.20**
225-m								62	67	68			75		**1.60**

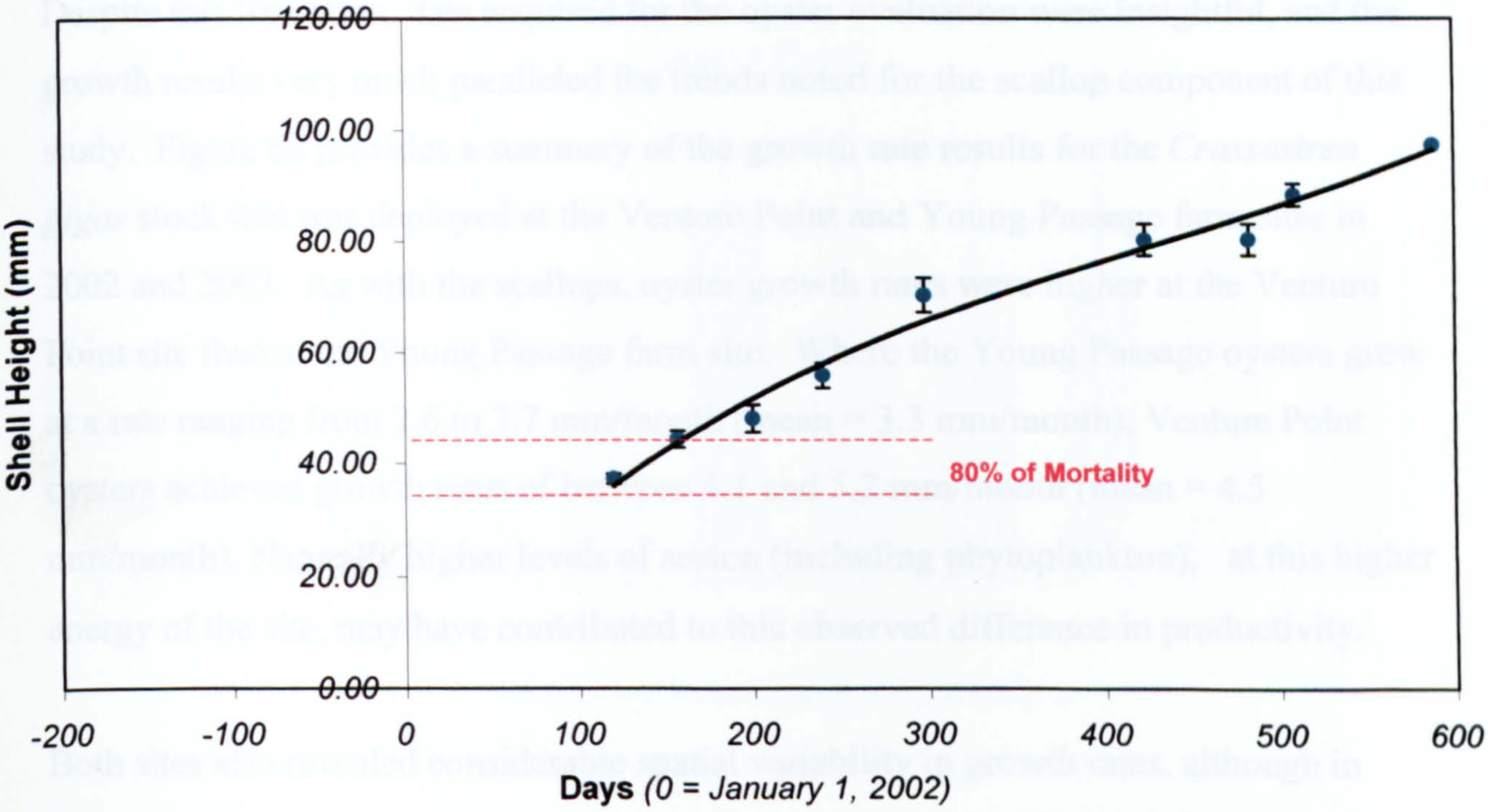
Figure 67 presents a similar evaluation of the scallop mortalities, for the Venture Point farm site, that occurred over the duration of the study. On average, overall mortality was approximately twice that reported for the Young Passage farm site at each of the downstream stations (mean = 5.48; SD = 1.54), yet there was no significant difference in mortality among these ten stations ($P<0.05$). Despite this significant difference between study sites, this mortality level remains well within the acceptable (and anticipated) range for the shellfish aquaculture industry in coastal British Columbia.

The growth data presented in Figure 67 (upper graph), were pooled from all site data (n=10 stations; means ± 95% confidence limits), and fitted with a polynomial curve to best describe the seasonal change in growth rate among these data. The mortalities incurred at this site are shown in the lower table, with the individual losses for each stations shown in the right column (expressed as a percentage of animals deployed).

Scallop mortality at the Venture Point site occurred primarily during the initial stages of the growth cycle, immediately after seed deployment, with 80% sustained in the scallops that were between 30 and 45 mm shell height (see upper graph, Figure 67) and over a very short period of time (approximately 45 days). As shown in the lower table, there was also no downstream station that incurred a greater level of mortality, nor a different pattern of mortality (scallop size range). In fact, all of this mortality was incurred at the very beginning of the growth cycles for both the shellfish and finfish at this site, with the latter comprised only of smolts.

Given the observed pattern of mortality at the Venture Point farm site, it is speculated that this was related to a combination of transportation and initial site acclimation effects for these bivalves. There is no evidence that any of these mortalities were related to any aspect of the finfish operation, i.e., organic and/or any constituent contaminant loadings.

Figure 67: Growth of Japanese scallop (*Patinopectin yessoensis*) at Venture Point farm site showing incurred mortality in test animals (lower table) and the period during the production cycle in which mortality occurred (upper graph, section identified in red). Growth curve shows measured shell heights (n=10) with 95% confidence limits; polynomial fit illustrates seasonal growth rate increases.



	Individual Sizes for Each Recorded Mortality (mm)																					% Mortality
0-m	33	38	39	39	39	39			40	40	41	42	47			52	55	62				**5.60**
10-m	31	35	36	37	37	38	39		40	40	41	44	45	45	45							**5.60**
20-m	30	32	33	36					40	43	55	61										**3.20**
30-m		35	36	36	37	38	38		40	54	60							67			72	**4.40**
50-m	30	36	38						40	40	41	42	45			51						**3.60**
75-m	30	35	35	37	39	39			40	40	40	41	45				59					**4.80**
100-m	28	31	31	32	35	35	36	39	39	39	40	40	41	42		53	60	62	65	66	67	**8.00**
125-m	32	34	37	37	38	39	39		40	40	40	43	45	48			58					**5.60**
175-m	30	35	37	37	38	38	38	39	41	41	41	41	42	42	44	54	58	64				**7.20**
225-m	30	31	31	31	36	37	38	39	40	41	41	41	43	43	44					68	77	**6.80**

6.4.2 *Crassostrea gigas* Growth and Survival

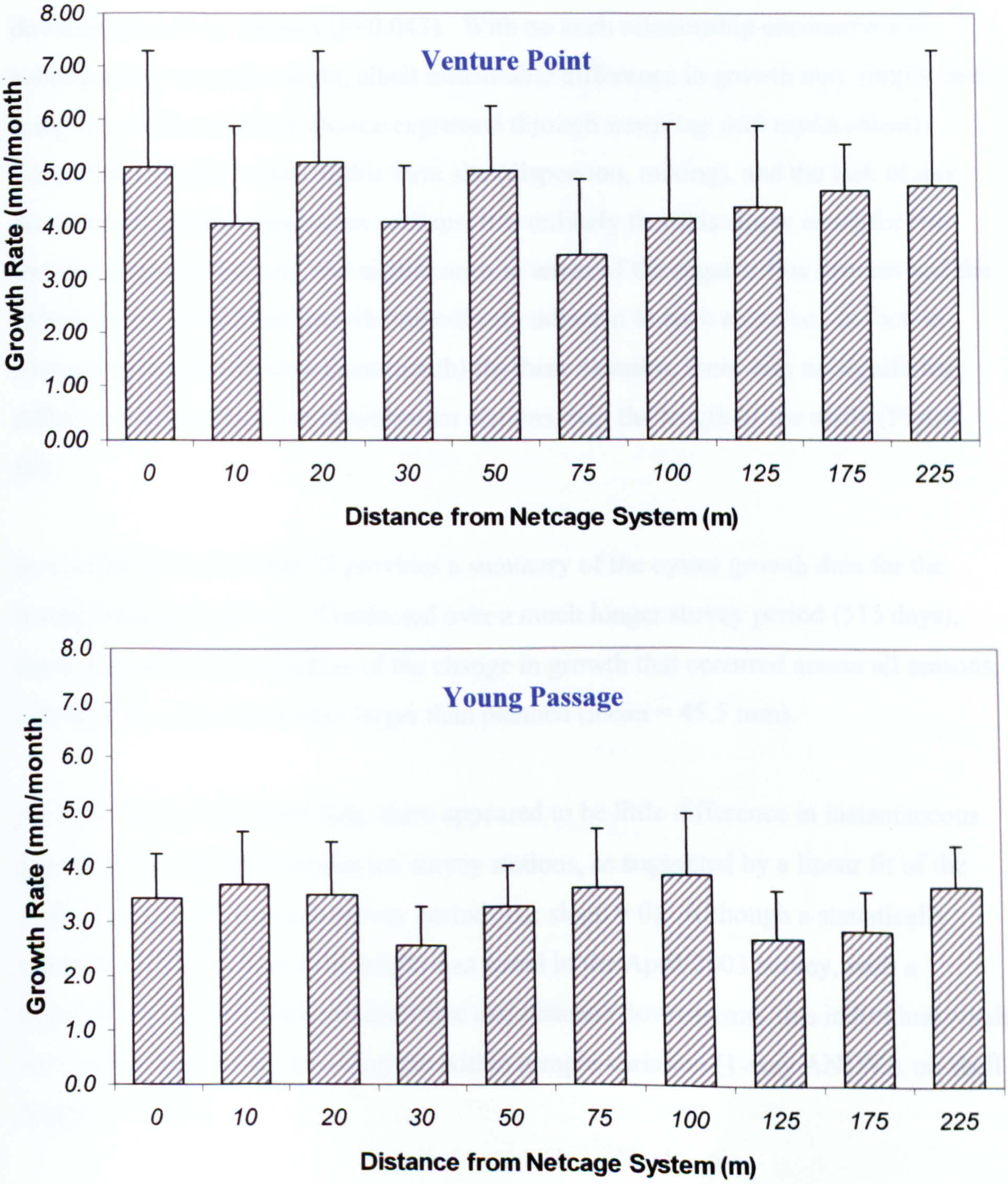
The timing for deployment of oysters in this study did not coincide with the scallop seed availability acquired from the shellfish industry, nor with the production of oyster seed from local hatcheries. Hence, the oyster shellstock that was deployed at each of the study sites was physically larger than planned, and as such did not provide a complete production cycle (including spat/juvenile culture period) over which to rigorously assess growth characteristics of this species at the two study sites. Delayed deployment for the Venture Point farm site resulted in only four post-deployment surveys, while the Young Passage site allowed an evaluation of culture performance using data from seven surveys.

Despite this limitation, data acquired for the oyster evaluation were insightful, and the growth results very much paralleled the trends noted for the scallop component of this study. Figure 68 provides a summary of the growth rate results for the *Crassostrea gigas* stock that was deployed at the Venture Point and Young Passage farm sites in 2002 and 2003. As with the scallops, oyster growth rates were higher at the Venture Point site than at the Young Passage farm site. Where the Young Passage oysters grew at a rate ranging from 2.6 to 3.7 mm/month (mean = 3.3 mm/month), Venture Point oysters achieved growth rates of between 4.1 and 5.2 mm/month (mean = 4.5 mm/month). Naturally higher levels of seston (including phytoplankton), at this higher energy of the site, may have contributed to this observed difference in productivity.

Both sites also revealed considerable spatial variability in growth rates, although in comparing these averages (1-way ANOVA) there was no significant difference ($P<0.01$) detected among the ten downstream stations for either of these study sites. Temporal variability, illustrated in the sample standard errors (Figure 68), was higher at Venture Point stations than at Young Passage. This too may reflect the nature of the environment at Venture Point, particularly with respect to its physical oceanographic and physiographic characteristics (see Chapter 3).

A closer examination of the acquired growth data was completed in an effort to establish whether temporal effects (seasonal or age-specific) were apparent for the oyster performance observed at either of the two study sites. Figure 69 summarizes the growth data for each of the four evaluation surveys conducted at Venture Point. These data illustrate shell height measurements (mean, sample standard deviation, 95% confidence limits, n=5) for samples taken at each downstream station over this short assessment period (184 days).

Figure 68: *Crassostrea gigas* growth rate (mm shell height/month) determined using data acquired during production cycle at the Venture Point (upper bar graph) and Young Passage (lower bar graph) study sites. Values for each station represent sample means (Venture Point, n=4; Young Pass, n=7) with associated standard error estimates.



The tabulated results, and the lower graphical representation, also show the relationship between oyster size (mm shell height) and distance downstream of the finfish aquaculture site for each sampling period. The statistical significance of the slope (H_o: slope=0) is shown for each of these sampling events, along with the associated probability. Although not a rigorous assessment, due to the limited number of surveys conducted during the study, these data did indicate that the first sampling period (February/2003) revealed a significant decrease in growth (shell height) along the downstream survey transect (P=0.043). With no such relationship encountered in subsequent surveys this slight, albeit measurable difference in growth may simply be a sampling artifact (random chance expressed through *sampling with replacement*). Given the energetic nature of this farm site (dispersion, mixing), and the lack of any downstream growth response in scallops, it is unlikely that this single result for the oyster component has any real significance in terms of the organic flux process and the enhancement in shellfish growth immediately adjacent to such a source. In fact, in examining the growth rates (mm/month) for these samples, there was no significant difference among these ten downstream stations over the length of the study (Figure 69).

In a similar format, Table 22 provides a summary of the oyster growth data for the Young Passage farm site. Conducted over a much longer survey period (515 days), these data allow an evaluation of the change in growth that occurred across all seasons, although the initial seed were larger than planned (mean = 45.5 mm).

As with the Venture Point data, there appeared to be little difference in instantaneous growth across the downstream ten survey stations, as suggested by a linear fit of the shell height means for each survey period (H_o: slope = 0). Although a statistically significant difference in shell height was noted in the April/2003 survey, with a negative relationship between shell size and distance downstream, this individual result was not apparent when assessing the within sample variation (1-way ANOVA on shell height; $P < 0.01$).

Figure 69: Venture Point oyster (*Crassostrea gigas*) growth data. Upper summary table includes mean size (mm shell height) with sample standard deviations and 95% C.L.'s for each survey period (4 total) identified as Julian Day (start Jan/2003). The lower graph tests these mean growth data for each period against a simple linear model, the significance for each shown in the upper table.

Day:	**59** (28 Feb. 2003)			**83** (3 Apr. 2003)			**134** (24 May 2003)			**184** (3 August 2003)		
	Mean	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*
0	93.60	2.07	1.82	99.80	3.63	3.18	99.00	8.54	7.49	104.20	6.22	5.45
10	91.60	1.95	1.71	96.40	4.83	4.23	96.40	7.77	6.81	99.80	5.72	5.01
20	91.80	2.68	2.35	97.80	2.17	1.90	104.00	11.96	10.48	99.60	9.48	8.31
30	90.40	2.07	1.82	95.80	8.23	7.21	99.40	13.01	11.40	103.80	8.58	7.52
50	88.00	6.28	5.51	94.40	9.24	8.10	101.00	8.97	7.86	110.20	11.92	10.45
75	87.00	6.04	5.30	92.40	1.14	1.00	97.20	7.56	6.63	104.00	3.46	3.04
100	93.40	3.78	3.31	95.00	6.36	5.58	100.20	8.11	7.10	103.40	6.88	6.03
125	89.20	4.32	3.79	95.20	3.19	2.80	105.20	6.98	6.12	106.20	9.26	8.11
175	88.60	4.67	4.09	94.00	5.70	5.00	99.40	7.60	6.66	108.20	5.59	4.90
225	85.80	2.59	2.27	95.20	3.96	3.47	94.80	2.39	2.09	101.20	2.17	1.90
slope	-0.023			-0.014			-0.009			0.009		
r	-0.646			-0.519			-0.214			0.192		
signif.	*	*P=0.043*		*n.s.*	*P=0.124*		*n.s*	*P=0.553*		*n.s*	*P=0.596*	

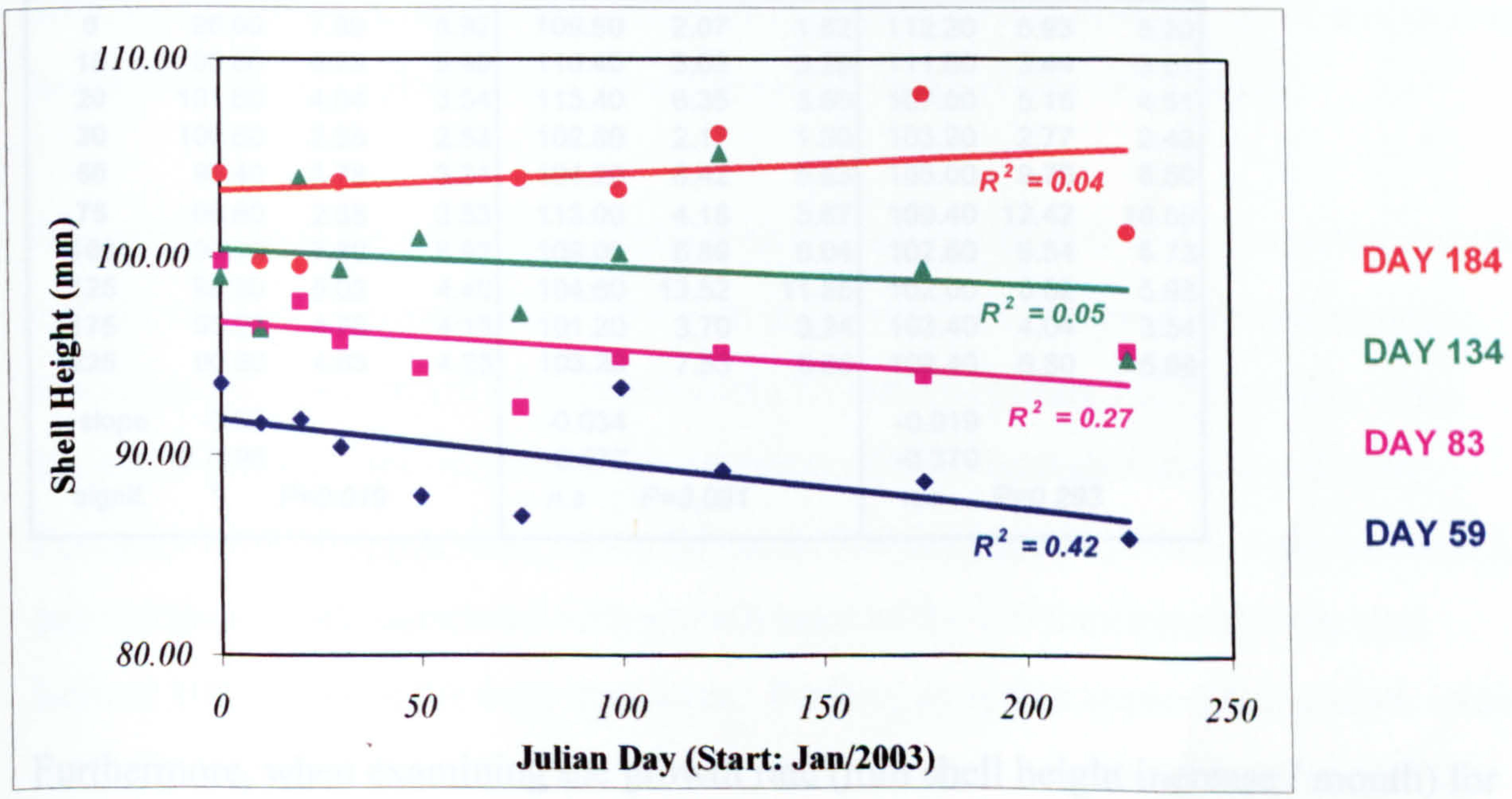


Table 22: Young Passage oyster (*Crassostrea gigas*) growth data. Summary table includes mean size (mm shell height) with sample standard deviations and 95% C.L.'s for each survey period (7 total) identified as Julian Day (start Jan/2002). The relationship between oyster growth and distance from the netcage system, for each survey period, was tested against a simple linear model; the significance for each is shown at the bottom of each column.

Day:	**118** (28 Apr. 2002)			**156** (5 Jun. 2002)			**199** (18 Jul. 2002)			**247** (4 Sept. 2002)		
	Mean	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*
0	55.80	3.90	3.42	58.80	0.84	0.73	60.80	3.27	2.87	63.80	3.56	3.12
10	53.20	1.30	1.14	58.60	2.70	2.37	56.80	1.64	1.44	64.00	4.74	4.16
20	50.20	5.59	4.90	58.20	2.68	2.35	63.00	4.24	3.72	64.00	2.92	2.56
30	54.20	3.56	3.12	56.80	3.42	3.00	58.60	5.03	4.41	63.40	1.14	1.00
50	51.40	2.51	2.20	59.20	1.92	1.69	61.00	2.45	2.15	68.00	2.24	1.96
75	55.20	3.56	3.12	60.00	5.24	4.60	60.80	3.03	2.66	65.20	2.86	2.51
100	53.60	2.70	2.37	62.40	4.22	3.70	57.40	2.51	2.20	62.80	4.15	3.64
125	55.60	3.36	2.95	57.60	2.51	2.20	61.40	2.51	2.20	61.40	2.51	2.20
175	53.60	2.30	2.02	59.40	2.41	2.11	61.80	1.64	1.44	61.60	4.56	4.00
225	51.40	2.70	2.37	58.60	2.70	2.37	63.00	2.35	2.06	64.00	2.65	2.32
slope	-0.0028			0.003			0.012			-0.009		
r	-0.1109			0.173			0.429			-0.354		
signif.	*n.s*	*P=0.760*		*n.s*	*P=0.633*		*n.s*	*P=0.217*		*n.s*	*P=0.316*	

Day:	**459** (4 Apr. 2003)			**520** (4 Jun. 2003)			**578** (1 Aug. 2003)		
	Mean	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*	*Mean*	*Stdev.*	*95% CL*
0	96.60	7.89	6.92	109.60	2.07	1.82	112.20	5.93	5.20
10	95.60	6.73	5.90	110.40	3.65	3.20	111.60	3.44	3.01
20	101.60	4.04	3.54	113.40	6.35	5.56	107.00	5.15	4.51
30	100.60	2.88	2.53	102.80	2.17	1.90	103.20	2.77	2.43
50	98.40	3.78	3.31	104.20	6.42	5.63	105.00	9.70	8.50
75	96.60	2.88	2.53	113.00	4.18	3.67	109.40	12.42	10.89
100	94.40	7.89	6.92	109.00	6.89	6.04	102.60	6.54	5.73
125	96.80	5.02	4.40	104.60	13.52	11.85	102.00	6.82	5.98
175	95.20	4.76	4.18	101.20	3.70	3.24	103.40	4.04	3.54
225	90.60	4.83	4.23	103.20	7.53	6.60	108.40	6.80	5.96
slope	-0.03			-0.034			-0.019		
r	-0.7198			-0.577			-0.370		
signif.	*	*P=0.019*		*n.s*	*P=0.081*		*n.s*	*P=0.293*	

Furthermore, when examining the growth rate (mm shell height increase / month) for the Pacific oyster (*Crassostrea gigas*) grown at Young Passage, the average rate of growth (across all survey periods; n=7) were not significantly different (1-way ANOVA; $P<0.01$) for any of the ten downstream stations. These growth rates varied little among the ten downstream stations, illustrated in the summary statistics (mean ± standard deviations) presented previously in Figure 68, with between-station and within-station variation of typically less than 1.0 mm/month.

Despite a lack of statistical significance in *Crassostrea gigas* growth rates among downstream stations, there does appear to be subtle evidence of growth enhancements that may be related to the proximity to the finfish system and to a source of supplemental organic material. Figure 70 presents an overview of the oyster growth rates at Young Passage. This colour contour plot shows rates of growth as the dependent variable, standardized as mm shell height/month, as a function of time (x-axis; Julian Day starting January/2002) and distance downstream of the finfish system (y-axis, meters). The growth rates are represented as a gradient, with zero-growth as dark blue and maximum growth (8 mm/month) indicated with the darkest shade of orange.

As with the scallop growth rate summaries, presented previously in this format, the seasonality in oyster growth is clearly portrayed in Figure 70. Periods A and B, corresponding to the spring (March-June), shows growth in the deployed oysters that, although variable in magnitude among stations downstream of the cage system, are predictably related to early phytoplankton bloom events of this coastal region. During the late fall and winter months, exemplified by Period C, growth rates are minimal, and often close to zero. Cooler water temperatures, reduced photoperiod, and the resulting lower food/phytoplankton levels would naturally support this observation.

However, despite the anticipated depression in shellfish growth over the winter months, a slight increase in growth rates were seen in the region adjacent to the finfish cages that extended downstream to approximately 75 meters (Figure 70, Period D). The increased growth rates occurred from September/2002 through January and early February/2003. Growth rates within this area, and during this period, were between 3.5 and 4.0 mm/month, compared with growth rates of 0 – 2.0 mm/month in the area beyond 100 meters of the farm structures. Further, as spring approached growth rates naturally increased the entire length of the study transect, masking the "process" that was enhancing growth in the near-field sampling stations.

Figure 70: Contour plot of *Crassostrea gigas* growth performance at Young Passage study site. Growth data standardized as change in average shell height (n=10; mm/month) across each of the 7 survey periods (X-axis; Julian Day starting at January/2002)) for each of the 10 longline stations (Y-axis). **A**: March-May, 2002. **B**: April-June, 2003. **C:** winter, 2002. **D:** late fall, 2002 through winter (into early 2003).

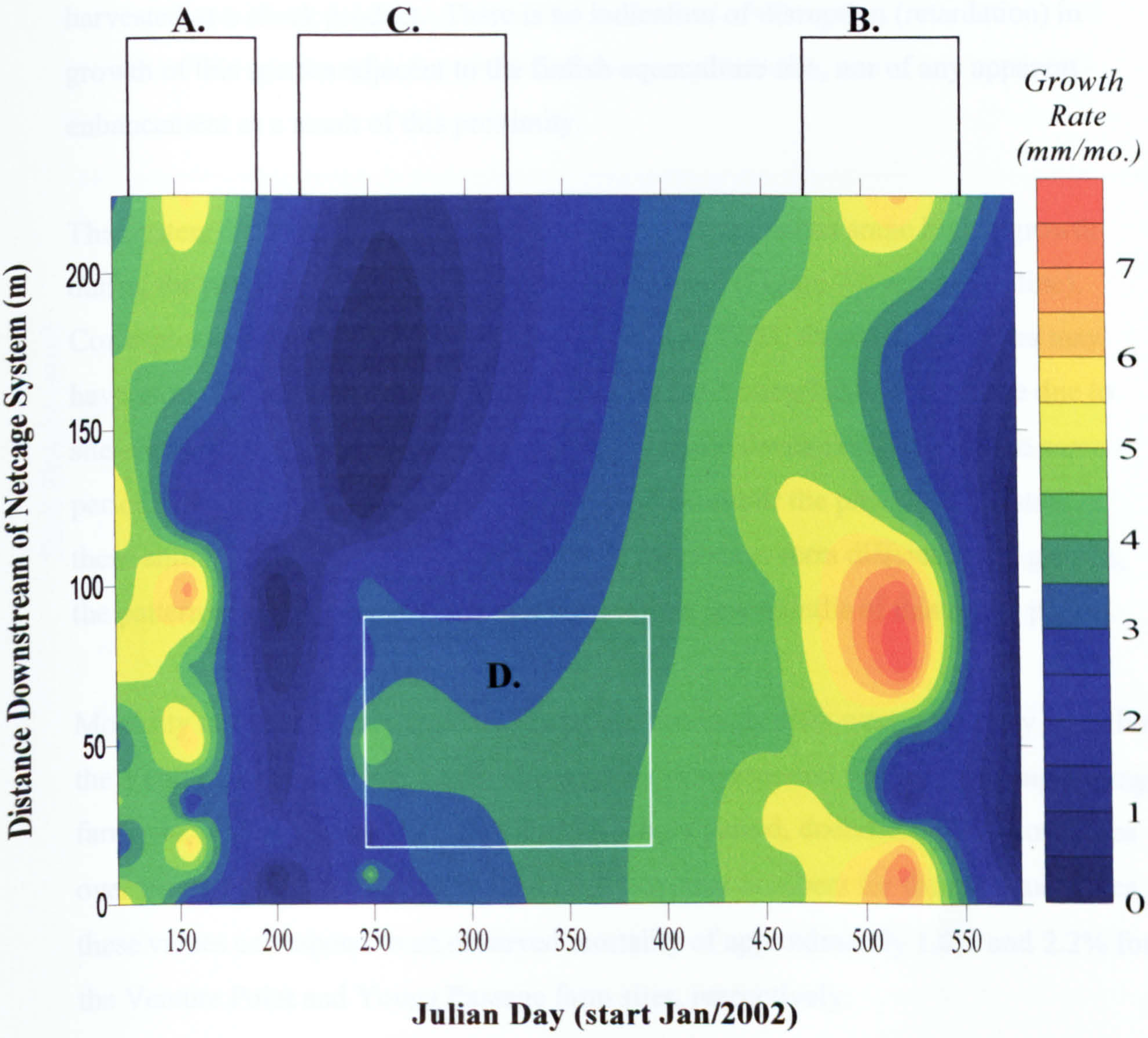


Figure 71 presents a direct comparison of the *Crassostrea gigas* growth curves for the Young Passage and Venture Point study sites. Given that the Young Passage site provided a substantially greater amount of growth data, with the extended monitoring period completed during this study, these results are employed as a basis of comparison for this growth. A 3rd-order polynomial is fit to the Young Passage growth data (blue dotted line), which represent sample means ± 95% confidence limits calculated from data acquired and pooled from all of the downstream stations (assuming no difference in growth among these records; discussed above).

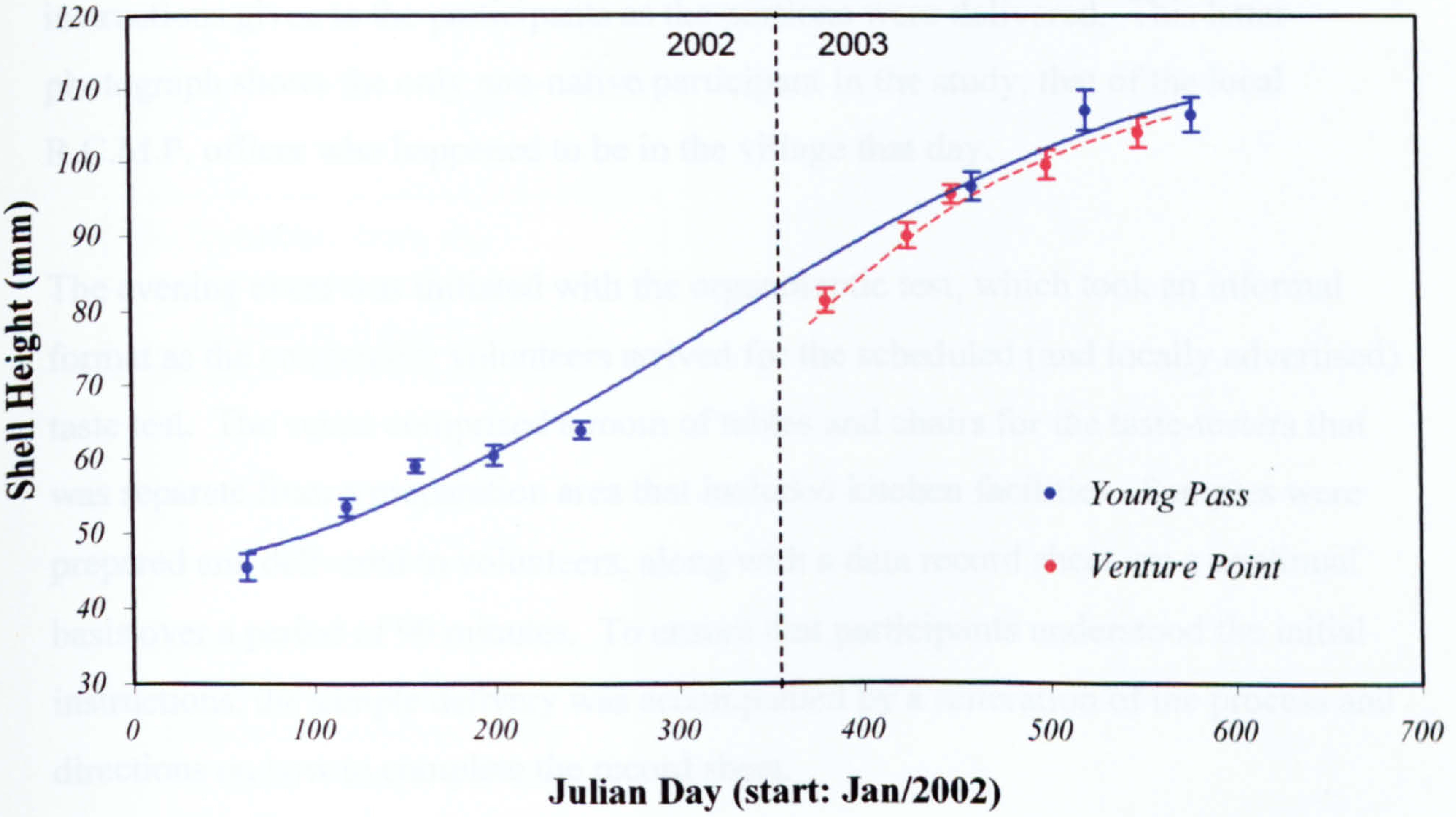
The oyster growth at Young Passage follows a typical logistic curve, with shell height appearing achieve a maximum at between 10 and 11 cm. These animals would be categorized as large, from a commercial perspective, and not atypical of the size harvested as a shuck product. There is no indication of disruption (retardation) in growth of this species adjacent to the finfish aquaculture site, nor of any apparent enhancement as a result of this proximity.

The oysters deployed at Venture Point appeared to experience some higher growth during the period immediately following deployment (Figure 71; red dotted line). Corresponding to the late winter and early spring of 2003, these larger oysters may have grown at a rate faster than that observed at the Young Passage farm site due to site-specific differences in seston availability (natural development), over the same period, or as a result of the differences in the sizes and/or the physiological state of these animals at this particular time. Despite these short-term differences in growth, the pattern of growth became comparable within a few months of this initial period.

Mortality rates for the oysters deployed at the two study sites were extremely low. For the Venture Point farm site a total of 6 animals were reported while the Young Passage farm site, which was monitored for a much longer period, documented 13 mortalities over the survey period. Given the initial deployment numbers for the two study sites, these values correspond to an observed mortality of approximately 1.0% and 2.2% for the Venture Point and Young Passage farm sites, respectively.

The mortality levels estimated for the two study sites were well below normal, and acceptable, shellfish industry levels (BCSGA, pers.com.). The low levels were most likely a reflection of the large sized animals that were deployed for the study, and as such did not include any evaluation of mortality that is generally inherent in the juvenile shellstock that is deployed at a shellfish farm. Nevertheless, the low mortalities reported for this study fully support those data documented for the scallop component, suggesting that the proximity of the finfish farm has no influence on shellfish survival at these sites.

Figure 71: Comparison of *Crassostrea gigas* growth curves for Young Passage (blue) and Venture Point (red) study sites. Data averaged across downstream stations (n=10), with 95% confidence intervals shown for each. Growth measurements for Young Passage span 18 months, and therefore are used to fit a 3rd-order polynomial model to these data. Venture Point results shown for short period in 2003, with higher initial growth rates (late winter and early spring) suggested by these data.



6.4.3 Organoleptic Test Results

The organoleptic test was conducted in October, 2003 with the assistance of 22 community volunteers of the Ka:'yu:'k't'h' / Che:k:tles7et'h' First Nation. The test was conducted at the community hall in Kyuquot Village, located on the northwest coast of Vancouver Island, and was incorporated into a traditional ceremony of thanks that included a community dinner.

Figure 72 provides a photographic summary of this study component. Photograph A shows the traditional drummers and singers performing prior to the event prayer and activities. Photograph B documents a First nation participant helping in the preparation of shellfish for the organoleptic tests; three women assisted in the preparations, one to shuck and size the portions, one to cook (separate skillets and dishware for test shellfish categories), and a final participant that helped assign and label portions according to the database designations. None of these individuals participated in the actual taste test component of the test.

Figure 72C and D show participants evaluating the shellfish portions. Photograph C illustrates the taste and data recording aspects, and Photograph D illustrates the verbal instructions given to the participants as the portions were delivered. This latter photograph shows the only non-native participant in the study, that of the local R.C.M.P. officer who happened to be in the village that day.

The evening event was initiated with the organoleptic test, which took an informal format as the community volunteers arrived for the scheduled (and locally advertised) taste test. The venue comprised a room of tables and chairs for the taste-testers that was separate from a preparation area that included kitchen facilities. Samples were prepared and delivered to volunteers, along with a data record sheet, on a continual basis over a period of 90 minutes. To ensure that participants understood the initial instructions, the sample delivery was accompanied by a reiteration of the process and directions on how to complete the record sheet.

Each of the participants was asked to complete three separate taste evaluations, for an anticipated total of 66 test results. Five of the returned data sheets were completed incorrectly (61, 62, 64-66) and were removed from the results spreadsheet. Given that these errors were produced from three individuals, with the latter three from the final participant entering the test, the elimination of these data were not considered to bias the assessment. The effects of participant were not tested, due to small sample size, but rather all responses were pooled for an evaluation of the main hypothesis, i.e., that there was no detectable difference in organoleptic properties of shellfish grown adjacent to a marine netcage facility as compared with shellfish grown well removed from such influences.

Figure 72: Organoleptic test conducted at the Ka:'yu:'k't'h' / Che:k:tles7et'h' First Nation community on northwestern Vancouver Island.

A. Traditional prayer and song conducted after test and prior to evening dinner.

B. Volunteer from the community assisted in preparing the test samples.

C. Taste testers included primarily members of the Kyuquot Village, but also drop-ins such as the local RCMP officer.

Table 23 summarizes the results of this simple organoleptic test. Of the 61 taste-test results, participants reported that they detected "no difference" among the samples for 18.03% of the responses where a difference actually was present. A total of 36.07% of the responses actually matched that of the allocated portions, although this was not statistically significant (t = -9.956; P< 0.001) and could not be attributed to anything other than random probability.

Table 23: Results of organoleptic test completed using scallops (*Patinopectin yessoensis*) grown at the Young Passage farm site. Data show results of 61 individual comparative tests of scallop tissue (1 from study farm against 2 from reference areas). The table shows actual position of sample versus that reported by test subject; a 0 indicates that subject could not identify a difference among the three samples.

Taste Test Record Number	Farm Scallop Test Position (actual)	Farm Scallop Test Position (reported)	Taste Test Record Number	Farm Scallop Test Position (actual)	Farm Scallop Test Position (reported)	Statistical Results
1	3	3	32	2	1	
2	1	3	33	1	2	Number of Subjects: 22
3	2	2	34	3	0	
4	2	1	35	2	2	Number of Tests (n): 61
5	2	1	36	3	3	Percent 0-Difference 18.03 %
6	1	1	37	2	2	Percent Match: 36.07 %
7	3	2	38	1	3	
8	3	1	39	3	3	1 = match; 0 = no match
9	1	2	40	2	3	
10	1	1	41	1	3	Probability of Matches Occuring
11	2	3	42	3	2	Other than by Random Chance:
12	2	0	43	1	1	P < 0.001
13	2	2	44	1	0	1-sample t; value=1
14	3	1	45	3	0	t = -9.956
15	2	2	46	1	3	
16	1	1	47	1	3	
17	1	1	48	3	3	
18	3	3	49	2	3	
19	3	0	50	2	2	
20	1	3	51	2	0	
21	3	3	52	2	0	
22	3	0	53	3	0	
23	1	1	54	2	2	
24	2	2	55	3	2	
25	2	1	56	2	1	
26	3	3	57	2	3	
27	3	2	58	1	0	
28	1	1	59	3	0	
29	1	3	60	1	2	
30	2	1	63	1	1	
31	2	3				

6.5 Discussion & Summary

This study component assessed both the potential for shellfish tainting (impacts on tissue palatability) and the co-culture effects on shellfish growth and survival. Assuming shellfish remain unaffected by water quality impacts originating from the finfish system (e.g., chemical residue persistence, see Chapter 5), in terms of either tissue burden levels of contaminants or by tissue tainting, then the potential effects of growth and survival on the resident shellfish becomes important considerations from a Multi-Trophic Aquaculture (MTA) commercialization perspective.

Shellfish Palatability:

Scallops grown in close proximity to two finfish facilities were evaluated for potential tainting of the consumable tissue, resulting from being cultured in waters containing measurable levels of organic waste originating from those finfish culture systems. The blind organoleptic tests revealed that 21 First Nation (and 1 non-native) test subjects could not detect any difference between scallops grown at these study sites compared with those cultured in remote, commercial shellfish aquaculture facilities. The animals used in this study component demonstrated that consumable tissues were not tainted, and presented normal taste, odour and texture responses when evaluated in this objective manner.

The importance of involving the First Nation people was considered critical to ensuring legitimacy of this study component, particularly in terms of social acceptance of the test results. The involvement of these volunteers in this study component has also allowed *direct* assessment of an environmental issue that has historically been offered as a concern to these coastal inhabitants (regarding environmental effects of salmon aquaculture) who rely on the wild fisheries (bivalves, fish) for their traditional subsistence and for commercial uses.

The use of community participants at each level of the study, including sample preparation, delivery, taste-testing, and data validation, further supports the benefits that could be realized through design of an objective (scientific) assessment that might otherwise garnish skepticism from social groups (e.g., fishers, recreationalists), or even from environmental activists. Resolution of many of the environmental issues that currently plague the salmon farming industry, and appear particularly acute in coastal British Columbia, could be addressed given such consideration. It is believed that the acceptance, and ultimate success, of using a rigorous scientific approach (e.g., appropriate hypothesis formulation; statistical survey design and data analysis; analytical methodologies) would also benefit significantly from *a priori* inclusion of representatives from such stakeholder groups.

The organoleptic test completed in this study focused on the scallop (evaluation of tainting effects on the consumable muscle tissue - *meats*), and thus the opportunity to provide a tainting assessment on whole-organism consumption was missed. Use of the oyster (had remaining numbers been sufficient for this study component) would have permitted an evaluation of a shellfish species that is typically consumed in its entirety. In this assessment, the contents of the gut (which could contain organics associated with the farm wastes) would presumably affect taste and/or texture of a prepared sample, and thereby indicate whether shellfish grown in proximity to finfish facilities.

Robinson et al. (*in prep.*) completed a similar organoleptic evaluation of shellfish grown immediately adjacent to a finfish aquaculture facility in New Brunswick, eastern Canada. They used mussels (*Mytilus galloprovincialus*) in their study, and given that this animal is consumed whole, the results provide important supplemental information to that acquired in this initiative using scallops. As with the scallop results, the Robinson et al. (2002) study revealed that no tainting effects could be detected in mussels grown adjacent to salmon aquaculture facilities. Further, given that the east coast study revealed a measurable increase in growth of mussels adjacent to the finfish test sites, one can assume that these animals were actively utilizing organic material from the farm and thus that these tissues were developed with a significant contribution of this seston component. If tainting (taste, odour, and/or texture) was a legitimate concern for finfish-shellfish co-culture, then this scenario would have provided the best opportunity for quantifying such a response.

The demonstrated absence of tainting in shellfish grown adjacent to marine finfish facilities, both in this study and in others (e.g., Robinson et al., 2002), represents a very important consideration in furthering the technical, environmental, and social dialogue regarding the development of this avenue of marine aquaculture. The participation of a coastal stakeholder group that has identified this issue as a major concern in the supporting the finfish aquaculture sectors (let alone considering any form of Multi-Trophic Aquaculture) further validates these data, and perhaps strengthens the argument for pursuing appropriate regulatory (policy) revisions that would legitimize such initiatives.

Shellfish Growth & Survival:

The culture performance of shellfish grown in close proximity to finfish culture facilities will provide a clear indication of whether integrated Multi-Trophic Aquaculture (MTA) could be a technically viable option for temperate latitude production operations. In theory, the release of organic material from the finfish component could become available to the shellfish, contributing to the natural seston levels and thereby enhancing growth of this component in an integrated finfish-shellfish system (Chopin et.al., 2003).

The growth and survival of the shellfish (*Patinopecten yessoensis* and *Crassostrea gigas*) deployed in this study indicated that there was no significant enhancement of shellfish culture performance regardless of proximity to the finfish culture system. Despite the flux of organic material from the farm site (see Chapter 4), growth rates remained consistent across all downstream stations, and these rates were comparable to those reported within the regional shellfish aquaculture industry.

The evidence suggesting that the organic wastes from the finfish system do not enhance shellfish growth adjacent to such a source of organics has not, however, been corroborated by the literature. In a recent study conducted by Chopin et al. (2003) in eastern Canada New Brunswick), their research documented a very significant increase in growth of both mussels (*Mytilus galloprovinciallus*) and kelp (*Laminaria saccharina*) cultured within the influences of a salmon net-cage system. In terms of the bivalve species evaluated, these animals displayed a growth enhancement estimated at 100% over that of the Reference area, clearly demonstrating the interactive effects of the salmon component with that of the co-cultured mussels.

The question becomes: *why would growth be enhanced in one area and not in another?* Three possible processes are presented herein to explain these results.

- Seston composition, concentration and the contribution/utilization of the farm-derived organic waste component;
- Dissolved nutrient flux and possible indirect effects on localized phytoplankton composition and concentrations; and/or
- Hydrodynamic attributes of the site and the effects of system infrastructure on bioavailability of the seston fractions.

Although the statistical evidence confirming that shellfish growth rates (and survival) were not significantly different (enhanced or constrained) as a result of culturing the shellstock (scallops and oysters) adjacent to a marine net-cage system in this study, there was some indication that growth rates were periodically greater immediately adjacent to the net-cage system than further downstream. This qualitative observation would suggest that the contribution of organic waste particles to the natural seston may contribute some, albeit minor (and periodic) growth within these shellfish resources and that this contribution may have a seasonal component.

The test oysters, in particular, revealed a slight elevation in growth rates at each of the farm sites in an area immediately adjacent, and downstream, of the salmon net-cage systems. This growth enhancement appeared to occur within approximately 50 meters downstream of the net-cage system, and only during the late winter months.

Effects of Seston Structure

It is speculated that most nonsiphonate shellfish species actively select their food from the seston fractions, and that the consumption of seston occurs in proportion to availability, or opportunistically (Coma et al., 2001). Scallops, and many other bivalve species, have been shown to have the ability to select particles from natural assemblages of seston, and to preferentially reject particles of poorer quality (Bacon et al., 1998; MacDonald and Ward, 1994; Wong and Cheung, 1999). In addition, these studies revealed that particle selection efficiency increased with POM concentration, and that for the sea scallop (*Placopecten magellanicus*) a reduction in clearance rate and production of pseudofaeces was used to regulate ingestion when particulate levels increased.

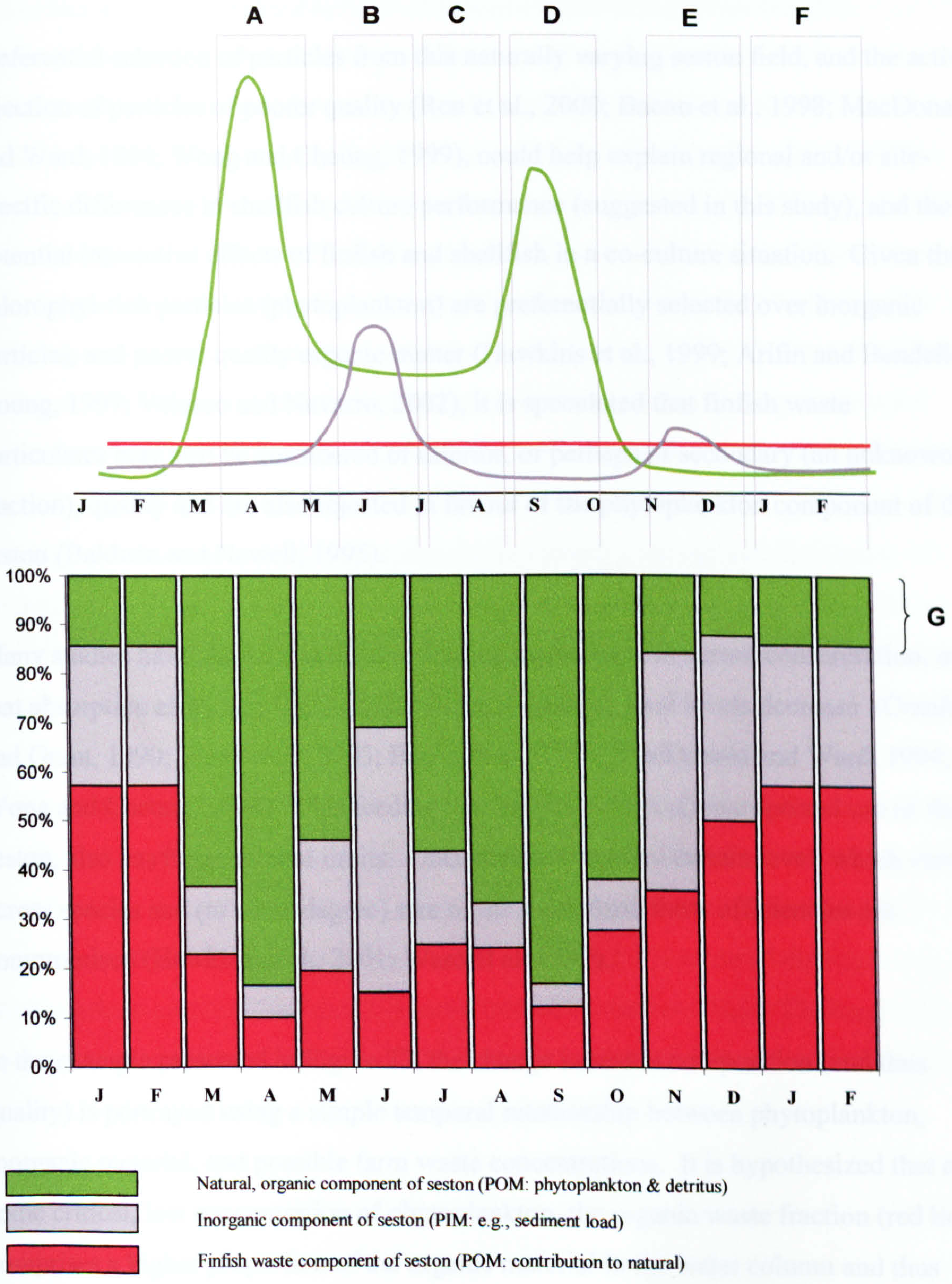
Many studies have examined the effects of increasing particulate concentration on feeding behaviour in bivalves, including scallops, mussels, oysters and clams (MacDonald and Ward, 1994; Bacon et al., 1999; Cranford and Gant, 1990). The natural variability (and availability) of suspended particulate food can be attributed to local biological and physical attributes, resulting from processes including that of phytoplankton bloom development, flocculation, erosion of soils, and re-suspension of sediments (MacDonald et al., 1998; Grant and Thorpe, 1991).

The concept of seston quality (composition), quantity (concentration), and the feeding characteristics (physiological and behavioral) of shellfish species are proposed as the basis for explaining site-specific (or regional) differences in the interactive effects of co-culturing shellfish and finfish. Figure 73 presents a hypothetical (and simplified) illustration of the seasonal change in seston concentration and composition for a typical Canadian west coast site. The upper line graph shows the relative change in concentration (e.g., mg/L, ug/g) of phytoplankton (green line), the inorganic particulate matter fraction (gray line), and a theoretical level of organic material contributed from an adjacent finfish net-cage system (red line). For discussion purposes, the latter is presented as an average (and constant) flux given that concentration of organic waste matter from a farm will vary through the production cycle that, in turn, could be initiated at a number of times over the year.

Figure 73 suggests that seasonal increases in phytoplankton will occur in spring (A), comprised of largely diatomaceous species, and later during the year (late summer and early fall; D) when the populations are dominated by dinoflagellates. These blooms contribute significantly to the seston composition, representing over 80% of the available organic-inorganic material in suspension, in this example (lower bar graph).

An increase in levels of inorganic material (e.g., silt-clay fraction) is typically associated with periods of steady rain (Figure 73, F), and is often substantially higher (E) following periods of minimal precipitation (D). The highest flux of inorganic material to the coastal waters is a result of the melting of the snow-pack, which occurs

Figure 73: Hypothetical temporal variation in seston composition and concentrations for British Columbia coast, showing POM (green line/bars) and PIM (gray line/bars) components in relation to potential organic input (average flux) originating from a finfish facility (red line/bars). Simplified relationship illustrates seasonal change in relative quantity of each component (upper graph), as well as the corresponding proportional change (quality) of the seston available to shellfish.



in late spring and early summer (B). The proportion of inorganic material within the water column will, during this freshet period, usually exceed that of the organic fraction (Thomson, 1982). The proportion of inorganic to organic fractions within the seston will also have an effect on bivalve feeding and on absorption responses (Cranford et al., 1998).

Preferential selection of particles from this naturally varying seston field, and the active rejection of particles of poorer quality (Ren et al., 2000; Bacon et al., 1998; MacDonald and Ward, 1994; Wong and Cheung, 1999), could help explain regional and/or site-specific differences in shellfish culture performance (suggested in this study), and the potential interactive effects of finfish and shellfish in a co-culture situation. Given that chlorophyl-rich particles (phytoplankton) are preferentially selected over inorganic particles, and poorer quality organic matter (Hawkins et al., 1999; Arifin and Bendell-Young, 1997; Velasco and Navarro, 2002), it is speculated that finfish waste particulates may also be considered of inferior, or perhaps of secondary (an unknown fraction), quality and are also rejected in favour of the phytoplankton component of the seston (Baldwin and Newell, 1995).

Many studies have shown that bivalve feeding increases with seston concentration, and that absorption efficiency will actually increase as these food levels decrease (Cranford and Grant, 1990; Ward et al., 2003; Bacon et al., 1998; MacDonald and Ward, 1994; Wong and Cheung, 1999). This feeding mechanism, which allows optimal use of the seston, also has it operational limits. Once seston exceeds a certain level, which varies across species and (to some degree) size of the individual, these efficiencies are compromised (Hawkins et al., 2001; Ward et al. 1999).

In the example presented in Figure 73, the change in seston composition (and thus quality) is portrayed using a simple temporal relationship between phytoplankton, inorganic material, and possible farm waste concentrations. It is hypothesized that at some critical, low concentration of phytoplankton, the organic waste fraction (red line) comprises a higher proportion of the organic material in the water column and thus becomes utilized, opportunistically, by the shellfish. If the proportion of organic waste material comprising the seston represents a significant component (in terms of concentration) in naturally 'clear' waters, then the introduction of shellfish to such areas could potentially exploit this organic fraction, introduced to the system through the finfish aquaculture facility, and perform better than might otherwise be expected under these naturally low seston levels (Gardner, 2000).

In this study, phytoplankton levels are assumed to remain well above this critical level, and in fact are supported by a number of bloom events that maintain elevated concentrations of the '*preferred*' organic fraction. However, on the Canadian east coast, growth rates in mussels were reportedly very high (Chopin et al., 2002), and results indicated that these growth effects were directly related to the adjacent finfish farm operation (and presumably the associated organic waste flux). Although data are not currently available to test the seston structure hypothesis, the clear difference between growth results in studies conducted in a similar manner may be attributable to such a dynamic. On the east coast, natural seston levels are generally lower than those on the west coast, particularly during the long winters when water temperatures are very low (0° and sub-0° C) and extended periods of enhanced primary productivity are limited. Gardner (2000) revealed that low levels of seston, and seston quality (phytoplankton composition), as possible factors limiting the natural distribution of mussels in New Zealand. He also suggested that areas that experience long-term periods with low seston biomass do not enjoy a positive energy balance, and thus will not support natural populations of these shellfish species.

In this study, decrease in phytoplankton concentrations during the winter corresponded to a decline in growth rates among the test bivalve species. The slight enhancement of growth within the near-field may be attributed to the use of the organic waste fraction to supplement the low phytoplankton levels only during this seasonal depression in phytoplankton composition (quality) and abundance (concentration). The length of such periods may, in other areas (or sites), determine to what degree the contributed organic fraction might be utilized.

DOM and Phytoplankton Enrichment

The observed shellfish growth rates were similar in terms of both average and within-station variability across all downstream stations at the low energy study site. It was assumed that a significant increase in bioavailable organic material, originating from the finfish system, would have resulted in a measurable difference in these growth rates, presumably with a significant negative correlation with distance from the finfish system.

If the introduced particulate organics (farm wastes) were not a contributing factor to the observed differences in shellfish growth adjacent to the finfish farm facilities, then the subtle elevation in growth (discussed above) might be an indirect consequence of localized impacts to the near-field phytoplankton community. It is speculated that a dissolved nutrient flux from the finfish facility may locally stimulate phytoplankton growth, and as a result account for enhancement of shellfish growth. Although nutrient flux was not measured, Toro et al. (1999) showed that phytoplankton abundance, biovolume and chlorophyl *a* concentrations (under similar temperature and salinity regimes) were significantly higher adjacent to a finfish farm site in southern Chile, and that oyster growth rate was also significantly enhanced within this near-field region around the farm.

The introduction of dissolved organic matter (DOM) on a continual basis over a finfish production cycle may result in the localized stimulation of phytoplankton populations during periods of nutrient depression (given other conditions for photosynthesis are suited for such growth). Further, at sites where phytoplankton concentrations naturally occur at low levels, perhaps as a result of long-term nutrient limitation, this enrichment process may explain the observed shellfish growth increases within the near-field region surrounding such finfish aquaculture facilities (Toro et al. 1999).

In contrast, at sites that naturally support a chlorophyll-rich environment this nutrient enrichment '*signal*' may not have a quantifiable effect on the resident phytoplankton populations. If nutrient levels do not represent a limiting factor for localized phytoplankton community structure, further enrichment with the farm-derived dissolved organics would not be realized through a similar, near-field enrichment effect.

The empirical data to support this hypothesis was not acquired directly through the present study, but some of the evidence can be used in support of this proposed enrichment process. For example, although a significant organic waste signal was not detected in the shellfish performance data downstream of either study farm, the bioaccumulation of farm-derived contaminants in shellfish tissues situated within 150 meters of the net-cage system were measured. These contaminants (e.g., antibiotic residues) are likely diluted and dispersed in a dissolved state, and if not associated (adsorbed) with the particulate organic waste fraction, would not result in a measurable (and otherwise expected) increase in growth.

A sustained abundance of phytoplankton, providing the preferred chlorophyll-rich environment required of all shellfish species, is typical of most coastal areas in western Canada. Given that these natural conditions infer that the phytoplankton community is not nutrient limited, would also preclude the possibility of realizing any enrichment effects associated with a dissolved nutrient flux from the finfish farm.

In eastern Canada, the study by Chopin et al. (2002) documented a substantial, and significant growth enhancement of both macrophytes and shellfish grown adjacent to a finfish aquaculture facility. It is speculated that a dissolved organic matter flux contributed to this growth, particularly as evidenced by the macrophyte component of the study, i.e., resulting in a 46% growth increase over that of the Reference area. If nutrient levels are typically low within these waters (Bay of Fundy), then these data may also support the process of nutrient enrichment, phytoplankton stimulation, and enhanced shellfish growth performance.

Hydrodynamic & Infrastructure Influences on Growth

Despite the slight spatial differences in shellfish growth observed in this study a consistent, and significantly higher growth (in both scallops and oysters) were documented at the high tidal energy site than at the lower, quiescent site evaluated. Oceanographic characteristics of a site can directly affect growth and survival in shellfish, and in fact represent key factors in defining areas that might support suspended shellfish aquaculture (Brown et al., 1990; Cross and Kingzett, 1991). Temperature and salinity regimes have been identified as important variables that will affect growth (Ward et al., 2003; Walne, 1972) but both were comparable between these study sites over the entire survey period, and as such would not adequately explain such growth differences.

However, and although not measured in this study, the localized concentration and bioavailability of natural seston may have simply been greater at the high energy site as compared with that of the low energy site. Alternatively, the hydrodynamic characteristics of the higher energy site could have contributed to the observed growth differential through transport, and surface enrichment processes (higher circulation through the channel, greater nutrient mixing, upwelling and organic material concentrating – see Chapter 3). Hawkins et al. (1996) documented a significant increase in mussel clearance rates in response to food availability that was associated with tidal exchange. In their study, tidal variation accounted for a range of seston concentrations from 10 to 90 mg total particulate mass per liter over the tidal cycle.

Large and/or small-scale flow velocities could explain the differences in shellfish growth between the two study sites, as well as within the near-field area potentially influenced by the introduction of organic wastes from each of the finfish facilities. At sites with higher current velocities and more complex bathymetric features, a more turbulent water column contributes to mixing of the food resources (seston) available to the shellfish. This turbulence may lead to localized patchiness in the food supply, explaining the spatially variable growth expressed at the higher energy farm site as compared with the slower, laminar-flow site.

Further, the netcage system itself may contribute to localized turbulence (mixing) patterns, and cause some upwelling or downwelling of water as it passes around the system and continues downstream along the shellfish longline. The apparent reduction in mean growth rates within the 10-30 meter range of the net-cages may reflect such processes at this higher energy farm site, particularly given that current flows at this site are largely unidirectional.

Ackerman (1999) reviewed a number of studies that examined the influence of velocity on filter feeding, and suggested that positive effects on filter feeding processes were observed under laminar conditions, while a negative relationship has been reported for turbulent conditions. The localized growth depression observed around the higher energy farm site may have resulted from a feeding impairment under such turbulent conditions (Eckman et al., 1989), while the downstream flows became increasingly laminar and failed to disrupt this feeding process, and in fact enhancing feeding under conditions of slow-moderate velocity and laminar flow (Wildish and Kristmanson, 1997; Kirby-Smith, 1972). Given the possible enrichment in food resources resulting from the vertical mixing at this site, a positive (albeit it spatially variable) growth response was observed in both species of shellfish (Wildish and Kristmanson, 1997).

In contrast to the higher energy site, the oceanographic characteristics of the low energy farm site were comparatively subdued, with slow and laminar tidal flows around and through the net-cage system. A lack of measurable turbulence, and the inherent bi-directional flows at this site would also ensure that seston fields remain relatively homogeneous, although with flows of sufficient magnitude to allow for food replenishment (Ackerman, 1999; Peterson and Black, 1987)). Given the inherently low flows at this site, it was expected that a positive effect on growth would be reported (Eckman et al., 1989; Calahan et al., 1989; Wildish and Kristmanson, 1997), following their proposed *continuous unimodal function* that relates feeding effect to flow velocity.

However, in this study the low energy farm site revealed significantly lower growth rates that that of the higher energy site, contradicting this laboratory-derived relationship. Assuming a comparable level of food resources between the study sites, the reduced growth performance at the low energy site could be a function not simply of flow velocity, but rather of tidal quiescence, infrastructure impedance, and the combined effects of these factors on food replenishment rates. It is speculated that extended periods of tidal quiescence (no flow) could reduce ingestion levels (if food is periodically depleted) and thus limit overall growth depending upon the magnitude of these effects. The ability to replenish food within culture enclosures (e.g., nets, trays) may also become an inhibitory factor under such conditions.

Implications to Multi-Trophic Aquaculture Development

Results of this study component have demonstrated that shellfish palatability is not negatively impacted as a consequence of culture proximity to a finfish aquaculture facility. Given appropriate and periodic testing of shellstock for possible contaminant loads, perhaps as part of a MTA management strategy, it is clear that the potential for developing integrated aquaculture should not be constrained by perceptions of possible seafood tainting.

Data from this study suggests that the siting of an integrated finfish-shellfish MTA system will determine if, or to what magnitude, shellfish growth performance benefits might be realized through such a co-culture system. The lack of shellfish growth enhancement in this particular study, contrary to that documented by other similarly designed research initiatives (Chopin et al. 1999; 2001), has spawned a number of theories regarding the processes (and environmental conditions) under which organic waste material from a finfish operation will stimulate shellfish growth within the near-field region of such a facility.

The hypothesis that seston quality and composition dictates when (or if) an introduced organic material might be consumed by a suspended shellfish component has important implications to a proposed finfish-shellfish integrated aquaculture development. First, in areas where phytoplankton concentrations are high, and stable throughout the year, the literature (as discussed above) would suggest that this waste fraction may not be accessed by these animals and as such would not necessarily contribute to a reduction of this waste component as anticipated from a true polyculture system. However, if this waste fraction is not (or is only periodically) utilized by the shellfish component of such a system, then any waterborne contaminants that may adsorb and be dispersed by such particulate wastes would not be readily available to the shellstock and thus would not represent a significant seafood safety risk to such a culture component (see Chapter 5).

The suggestion that growth stimulation may occur as a result of the dissolved nutrient flux will also affect site-specific culture performance. In areas where nutrients are not limited, and phytoplankton populations are sustained at levels that allow maximum growth of shellstock (given due consideration of stocking densities), then a growth enhancement (benefit) would not be anticipated of an MTA system. If, however, a site is characterized by prolonged periods of nutrient deficiency, that effectively limits primary productivity, then a farm-derived dissolved nutrient flux may locally stimulate phytoplankton growth and hence increase the levels of preferred food resources for a deployed shellstock component of an MTA system. Given this scenario, increased shellfish growth would be expected within a near-field region of the finfish system defined by nutrient flux, dilution and dispersion processes, water column properties (light attenuation, temperature, salinity), and the assimilative capacity of the resident phytoplankton community. The potential transfer of contaminants that adsorb to particulate organic wastes would also be avoided given this interaction scenario, further supporting the potential of MTA from a seafood safety perspective.

It was also speculated that farm infrastructure, including shellfish growout apparatus and finfish system configuration/positioning, could affect site-specific growth potential given the interaction of these systems with varying hydrodynamic characteristics. Development of upwelling and/or down-welling flows around these structures could affect distribution and potential availability of food resources, while inclusion of shellstock within enclosures (e.g., nets, trays) could result in small-scale dynamics that could preclude access, and/or effective use, of suspended particulates.

Although the shellfish performance component of this research project did not reveal significant growth enhancements when cultured in close proximity to finfish, there were no negative consequences of such an integrated aquaculture approach. Given the potential of growth benefits, and the theories describing the processes that may govern site-specific performance, *integrated*-MTA remains a realistic goal in terms of commercialization and from a seafood safety perspective.

CHAPTER 7

General Discussion and Conclusions

7.1 Introduction

The integration of multiple species into a common aquatic food production system, i.e., polyculture, has been employed in Asia (and particularly China) for many centuries. The practise of polyculture has resulted in the design and operation of systems that have demonstrated increased levels of production over that of monoculture systems (Behrends, et.al., 1985; Pavel, et.al., 1985; Perry and Tarver, 1987), with the majority of polyculture systems developed for freshwater, brackish water, and seawater ponds.

The most productive, and well-studied, polyculture systems include combinations of fishes, or fishes with shrimp/prawns (Perry and Tarver, 1987; Dos Santos and Valenti, 2002; D'abramo et.al., 1986; James et.al., 1988; Scott, et.al., 1988; Garcia-Perex, et.al., 2000). Tilapia (*Oreochromis niloticus*) and the freshwater prawn (*Macrobrachium rosenbergii*), in particular, represent the most widespread aquaculture species, having globally produced in excess of 800,000 tons and 130,000 tons in 1998, respectively (FAO, 2000). These species, representative of different trophic levels, have also provided the best combination of species for polyculture purposes; both species reach commercial size within 5 months, both express optimal growth within the same temperature regime, and both can tolerate low water quality (New, 2000).

In a polyculture system the tilapia uses the water column within the pond, relying on introduced feed and midwater productivity, while the prawn inhabits the epibenthic environment, and as a detritivore consumes introduced organic material depositing upon, or produced on, the benthic surface. Monoculture of the prawn leaves the water column underutilized, while culture of only the tilapia would not take advantage of the epibenthic space available for culture purposes (Dos Santos and Valenti, 2002). The combination of spatial separation, different trophic niches (non-competitive), and similar water quality requirements for achieving optimal growth, makes this combination of species an excellent choice for polyculture, and a good example of this form of aquaculture (Zimmermann and New, 2000).

Polyculture of marine species is much less developed, and established systems are confined primarily to land-based facilities that pump seawater and contain the various polyculture components in controlled quantities/biomass and culture conditions. These integrated systems, whether comprised of tanks, raceways, or ponds, are configured to balance the system such that the effluent from one component (e.g., finfish) is used by secondary (e.g., shellfish) and potentially by tertiary species such as seaweeds (Shpigel et.al., 1993a, 1993b). Shrimp, fish, and clams (*Tapes decussates*) have also been assessed, in terms of polyculture performance, in a pond system (Puigcerver, 1996). Other studies have assessed the potential of individual species to serve as polyculture candidates, determining (for example) nutrient uptake rates of giant clams grown in aquaculture effluent (Sparis et.al., 2001), or growth of milk fish (*Chanos chanos*) under monculture versus polyculture (with *Panaeus monodon*) conditions (Nammalwar and Kathirvel, 1988).

Marine polyculture has also been assessed, albeit through small-scale experimental studies, for open seawater systems. The culture performance of the Pacific oyster (*Crassostrea gigas*) has been evaluated as a co-culture species with the Pacific Chinook salmon (*Oncoryhnchus tshawytscha*), indicating that a significant increase in growth rate and tissue condition was apparent when this shellfish species was grown within the net-cages supporting the salmon (Jones and Iwama, 1991). The commercial potential of such results was suggested, but no further assessments completed in coastal British Columbia.

Scallops (*Placopecten magellanicus*) have also been evaluated as potential co-culture candidates with Atlantic salmon (*Salmo salar*). In the northeastern United States studies have indicated that growth and survival of this commercially important scallop at salmon farm facilities were not significantly different than that of monoculture operations for this shellfish species (Gryska et.al., 1996; Parsons et.al., 1999).

Initiatives on the east coast of Canada (New Brunswick) have recently evaluated the performance of mussels (*Mytilus trossolus*) and large macrophytes (*Laminaria*) cultured within the infrastructure of an open net-cage salmon (*Salmo salar*) aquaculture facility. Chopin et.al. (1999, 2003) and Robinson et.al. (2003) demonstrated that these integrated species perform significantly better within the influence of the salmon net-cage systems as compared with a monoculture arrangement removed from the apparent effluent effects of the finfish system. In contrast, a study in Tasmania, Australia (Cheshuk et.al., 2003) indicated that mussels (*Mytilus planulatus*) grown within 70 meters of a salmon (*Salmo salar*) farm revealed only very minor improvements in growth (shell height) and condition over the 14-month grow-out period. Stirling and Okumus (1995) also showed slight increases in mussel culture performance, grown at two salmon farm sites in Scotland, and suggested that enrichment of the seston field by organic material from the salmon farm was likely contributing to this observed elevation in growth.

Despite the apparent contradiction in shellfish performance results within potential finfish-shellfish integrated aquaculture systems (no difference or slight increases in growth), few scientific studies have shown or suggested any negative effects associated with the co-culture of marine finfish and shellfish species. While the present research has indicated that water quality effects from the salmon culture component can potentially have a negative effect on shellfish tissue quality, in terms of possible seafood safety, these measurable effects have distinct spatial and temporal attributes that could be managed within the operational framework of a finfish-shellfish integrated aquaculture system. Other studies (Bjoersholet.al., 1999; Nordtug et.al., 1999) have suggested that the potential for disease transmission must also be considered in developing appropriate management tools for integrated finfish-shellfish aquaculture, presenting some evidence that scallops can accumulate and excrete *Aeromonas salmonicida* subsp. *salmonicida* for 14 days after challenged with the bacteria. These laboratory studies also demonstrated that the salmon anemia virus (ISA) could not be retained in this manner.

The concept of polyculture has traditionally involved a combination of species that do not compete for available food resources, yet individually retain commercial value as aquatic food products. However, the term *polyculture* has also, more recently, been applied to any combination of species that may be co-cultured. In particular, the co-culture of various bivalve species (Nune et.al., 2003) are discussed in the context of polyculture, despite the fact that the two species are representatives of the same trophic level and thus compete for the available food supply.

To accurately reflect the intent of a true polyculture system, the phrase Multi-Trophic Aquaculture (MTA) is proposed. This term inherently defines the requirement of niche separation among the co-cultured species, while describing a structure that would intuitively combine species that would benefit directly from energy and/or organic material transfers between such trophic levels. Chopin and Yarish (1999) support the process by which selected polyculture species necessarily benefit from such linkages, and suggest that a macrophyte (algal) component is essential to any polyculture system to capitalize on the dissolved nutrient component of the system.

Results of this research would suggest that two options are available for developing MTA in coastal temperate waters, i.e., an *integrated* MTA system and/or an *adjacent* MTA system. Figure 74 illustrates the fundamental difference between the two forms of Multi-Trophic Aquaculture. In this example, 3 trophic levels are considered for a potential temperate MTA operation: (i) finfish, such as salmon; (ii) macrophytes, including kelps; and (iii) bivalve molluscs such as oysters, scallops or mussels. Figure 74A shows a possible integrated infrastructure configuration for these three MTA components, while Figure 74B presents an option for adjacent MTA using the same trophic level representatives.

The first approach endorses true multi-trophic aquaculture, thereby enabling the integration of various trophic level species within a single culture system (i.e., *integrated*-MTA or *i*-MTA). This approach recognizes the potential for water quality interactions that may originate from the finfish component of the system, and as such could be designed to take full advantage of these interactions, so as to benefit through optimal individual component production, while assimilating wastes of the finfish component and thereby reducing the environmental impacts of this culture activity. An *integrated*-MTA that incorporates a finfish component would also require recognition of the water quality interactions that could temporally (and differentially) affect food quality/safety in the other resident culture stock(s), whether shellfish, macrophytes, or another trophic level candidate for co-culture.

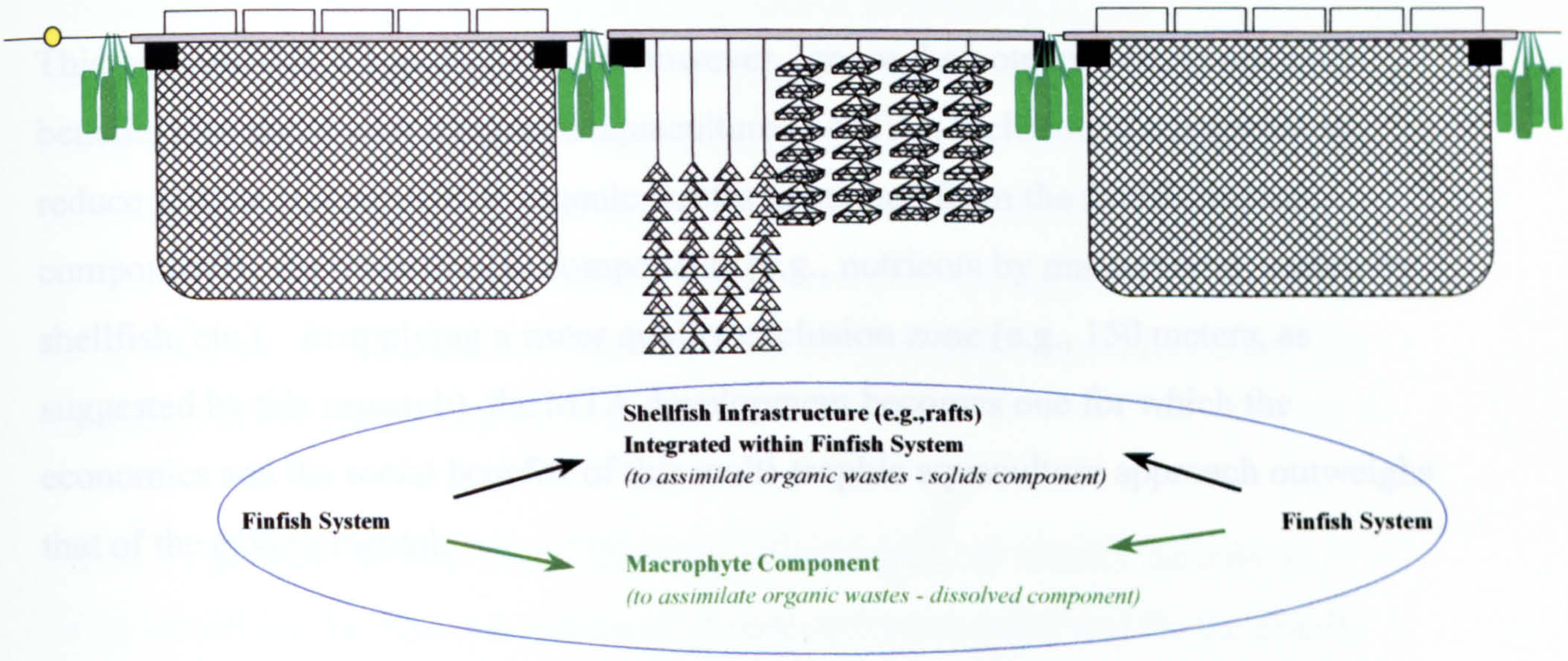
This research initiative has indicated that such interactive water quality effects are largely site-specific, with the occurrence and magnitude of these waterborne contaminant effects on an integrated shellfish component determined by the hydrographic characteristics of the *i*-MTA farm site, by the farm infrastructure configuration (that will affect the dispersion and dilution processes of waterborne contaminants released from individual netpens), and by the operational status of the farm (e.g., production levels, frequency and nature of chemotherapeutant treatments, use of copper-based antifoulants on nets, etc.).

The intermittent nature of waterborne contaminant impacts on *i*-MTA product quality may, as a mitigation measure, require implementation of a farm-based management (monitoring) system to ensure that proposed harvest schedules avoid periods during which *i*-MTA culture stock tissues experience elevated levels of the finfish component residues. Routinely conducted on all finfish exposed to feed additives that would, once elevated above specified levels/standards, be regarded as unsafe for human consumption, such additional product management protocols (for shellfish and for other MTA components) may not be regarded as any more cumbersome than regional industry-government management programs associated with bacteriological and/or biotoxin monitoring. These initiatives might require farm operational records for chemotherepeutant usage, projected clearance periods (fish, shellfish), sample collection (tissues; lot testing), regulatory harvest approvals, or other such monitoring data that may be considered appropriate to ensure product safety.

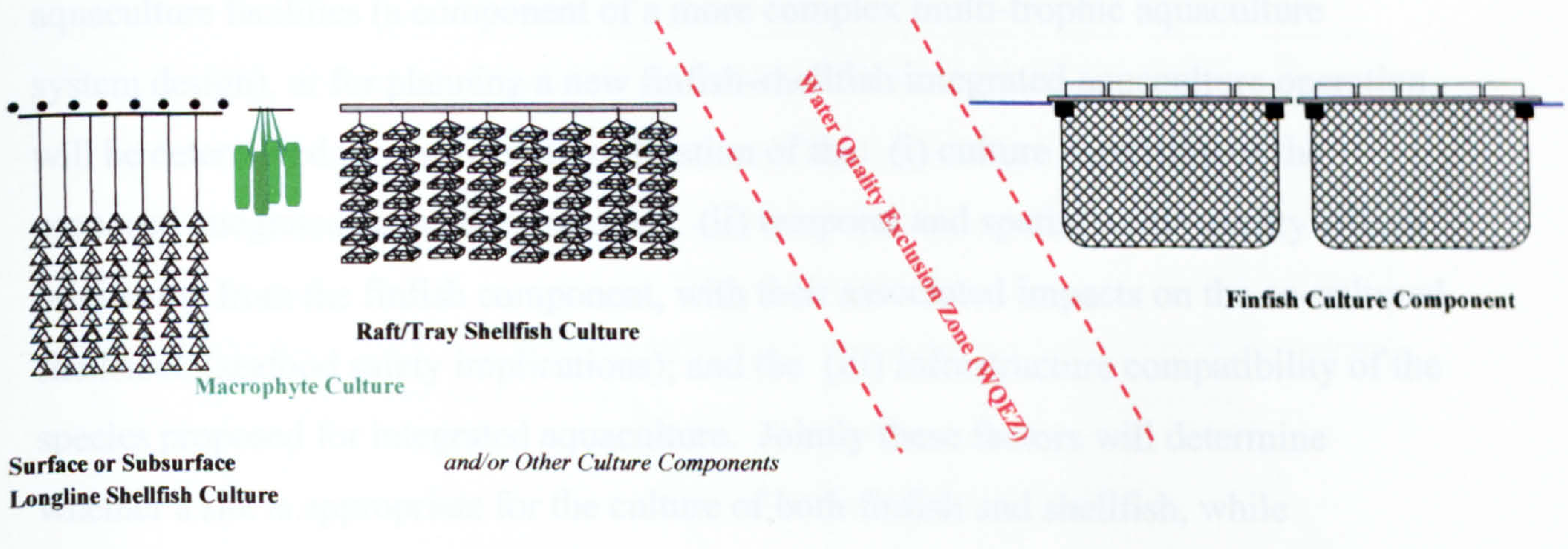
The environmental and resulting seafood safety risks associated with integrating other trophic level species with finfish (such as salmon), although considered low and manageable, would necessarily require change to regional Regulations and to operational Policies that would consider such risk and establish appropriate safe-guards for each of the MTA product components. The nature of such regulatory change will vary among jurisdictions, but compliance with established international trade standards for such products will likely dictate a degree of commonality among such initiatives.

Figure 74: Diagrammatic representation of the two proposed Multi-Trophic Aquaculture (MTA) approaches. **A**. an *integrated* system comprised of finfish, shellfish and macrophyte culture components; **B**. an *adjacent* system portraying the same components given a Water Quality Exclusion Zone (WQEZ).

A. *integrated-* MultiTrophic Aquaculture (*i-* MTA) System; 3-level



B. *adjacent-* MultiTrophic Aquaculture (*a-* MTA) System; 3-level



The second MTA option, in avoiding the potential for any waterborne contaminant interaction (albeit site-specific and temporal in nature), is to consider implementation of an appropriate, water quality exclusion zone (WQEZ) around the finfish component of the multi-trophic system (Figure 74B). In essence, the culture of shellfish, and co-cultured species representative of other trophic levels, would not be permitted within the area considered at risk from waterborne contaminants originating from the finfish component of the MTA. As a result, exposure of these culture stocks to these known waterborne contaminants would be eliminated, and the risk of accumulation and hence for biomagnification of such contaminants within their tissues would presumably be avoided.

This *adjacent*-MTA approach would, however, ignore the potential environmental benefits associated with integrated aquaculture (*i*-MTA), including the potential to reduce soluble and particulate organic wastes generated from the finfish system component by the other trophic components (e.g., nutrients by macrophytes, seston by shellfish, etc.). In applying a water quality exclusion zone (e.g., 150 meters, as suggested by this research), the MTA development becomes one for which the economics and the social benefits of this multi-trophic aquaculture approach outweighs that of the environmental.

7.2 Potential for Finfish-Shellfish Integrated Aquaculture

The potential for integrating shellfish aquaculture into existing marine finfish aquaculture facilities (a component of a more complex multi-trophic aquaculture system design), or for planning a new finfish-shellfish integrated aquaculture operation, will be determined through due consideration of the: (i) culture capability of the proposed integrated aquaculture species; (ii) temporal and spatial water quality effects originating from the finfish component, with their associated impacts on the co-cultured shellstock (seafood safety implications); and the (iii) infrastructure compatibility of the species proposed for integrated aquaculture. Jointly these factors will determine whether a site is appropriate for the culture of both finfish and shellfish, while determining the physical area within an authorized (licensed) aquaculture polygon (e.g., tenure, foreshore lease, concession) that is potentially available (biologically and legally) for integrating shellfish (or other) culture species.

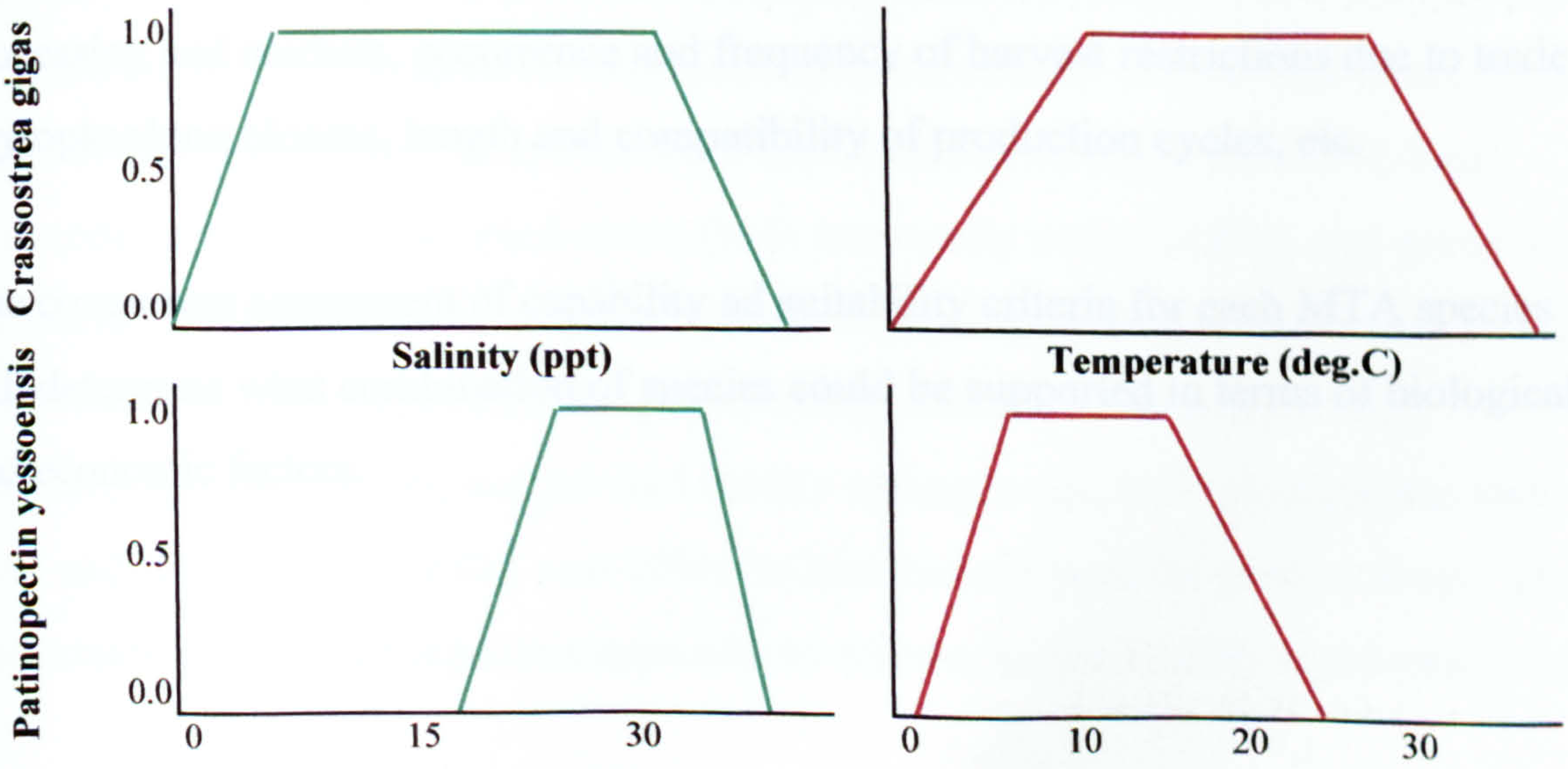
Each of these factors could also negatively affect the potential of a proposed *integrated* (or *adjacent*) multi-trophic aquaculture development, with the magnitude of any such limitation determining the technical and/or the economic viability of the proposed development.

7.2.1 Aquaculture Capability Considerations

The development of any aquaculture facility, be it is a hatchery, juvenile rearing, production/on-growing site, or whether it is marine or freshwater, will need to consider the biophysical requirements of the species proposed for culture, ensuring (as much as possible) that these factors are optimal, and that they thereby provide the greatest potential for maximum sustainable growth and survival of the selected aquaculture species. Such biophysical characteristics will determine the aquaculture <u>capability</u> for the species, by defining the tolerance range for the parameters that will directly (or indirectly) affect growth and survival.

Figure 75 provides an example of the aquaculture capability criteria (salinity and temperature) for the Japanese scallop (*Patinopecten yessoensis*) and for the Pacific oyster (*Crassostrea gigas*). These graphs show the optimal ranges for each parameter, as well as levels at which growth (and survival) would be compromised.

Figure 75: Example of shellfish capability plots for two physical parameters (temperature in red, salinity in green). Y-axis represents capability scale (0.0 = poor; 1.0 = optimal). Optimal culture range shown by upper plateau for each species/parameter.



In developing Multi-Trophic Aquaculture (MTA), site selection becomes increasingly complicated (and perhaps limiting) as multiple species, each with differing biophysical requirements, are proposed for such a facility. The ideal culture conditions will vary among species, and MTA site selection (or development at an existing aquaculture operation) will most likely require sacrificing (to some degree) the optimal site characteristics for at least one of the system components.

In introducing shellfish to an existing finfish aquaculture operation, for example, the biophysical attributes of the site would determine which species could actually grow, and survive, at the site. If salinities were high (29-31 ppt) and temperatures low (7-11° C) over the year, the site might be best suited to the co-culture of scallops rather than oysters, despite the fact that oysters could tolerate such environmental conditions. Alternatively, if the site experienced considerable seasonal fluctuations in salinity or prolonged, elevated summer temperatures (e.g., 15-18° C) then the selection of scallops could not be considered as these biophysical factors are outside of the tolerance range for the species.

Given that aquaculture is a commercial proposition, the decision to integrate species would be determined not simply by the biophysical criteria that would indicate whether the culture species would grow, and survive, at a site. Despite *less than optimal* biophysical conditions (for one or a number of the proposed MTA species), the evaluation would need to consider whether or not the environment could support an economically viable production operation, given that any sub-optimal characteristics could be mitigated through operational modifications and/or through adaptation of specific husbandry practices. Such suitability issues might include proximity to processing and markets, occurrence and frequency of harvest restrictions due to toxic phytopkankton blooms, length and compatibility of production cycles, etc.

The concurrent assessment of capability ad suitability criteria for each MTA species will determine what combination of species could be supported in terms of biological and economic factors.

7.2.2 Temporal and Spatial Water Quality Considerations

Four factors, related to the potential effects of waterborne contaminants (assessed, in part, within this research initiative), will play an important role in determining the appropriate location and configuration for suspended shellfish infrastructure within the operational boundaries of a marine finfish aquaculture facility, and as such will define whether and how an *adjacent* or an *integrated* MTA approach is deployed at a site. These factors include the:

(i) spatial extent of waterborne contaminant dispersion from the finfish aquaculture facility, determined through the oceanographic and physiographic attributes of the site;
(ii) loading characteristics of waterborne contaminants originating from the finfish operation (e.g., frequency, pulse duration);
(iii) shellfish uptake and retention dynamics (variable by species, age, environmental conditions); and the
(iv) regional regulatory environment that would establish the constraints to the co-culture of shellfish and finfish given the seafood safety risks inherent in MTA (particularly with a fully integrated production model).

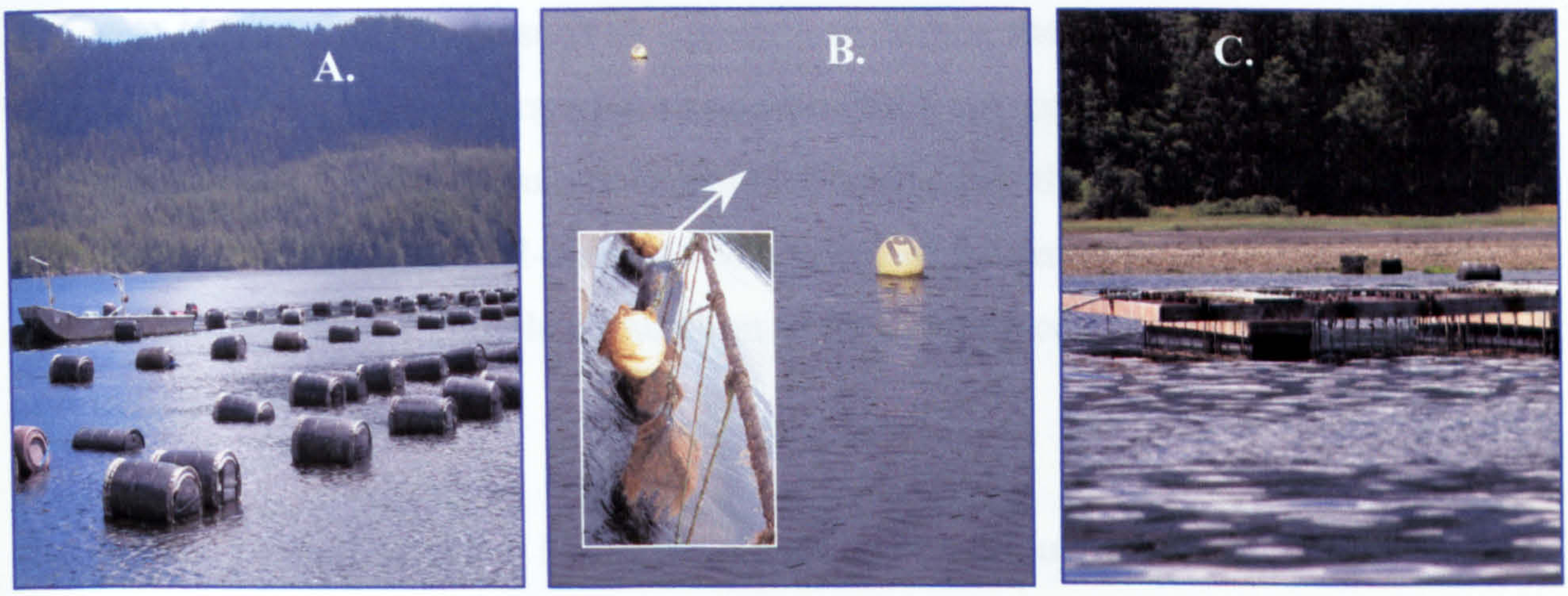
The empirical data acquired as a result of this research has demonstrated that *adjacent*, or *integrated* finfish-shellfish MTA represents a viable commercial option from the shellfish contamination risk, and thus the seafood product safety, perspectives. Waterborne contaminants originating from finfish production facilities do occur, they are quantifiable, and their dispersion pathways do suggest a measurable exposure risk to nearby shellfish resources (both to wild populations, that may occur in very close proximity to the finfish infrastructure, and to any introduced, commercial aquaculture shellstock placed within 150 meters of the finfish infrastructure). However, these contamination risks remain predictable (both temporally and spatially), and given due consideration of site-specific oceanographic and physiographic characteristics, as well as of finfish operational factors, the proper integration of finfish and shellfish aquaculture infrastructure, and product quality management, will minimize any such risks and ensure a compatible, cost-effective harmonization of the product lines. The application of a risk management approach to address seafood safety considerations is a prudent method by which the social and economic benefits of integrated aquaculture can be realized.

7.2.3 Infrastructure Considerations

The integration of a shellfish component within a marine finfish operation will also depend upon whether an *i*-MTA or an *a*-MTA approach is employed at the farm site. Where *adjacent* shellfish culture would occur outside of a Water Quality Exclusion Zone (WQEZ) and simply exploit the suitable, un-used space within a finfish aquaculture tenure (but remaining physically separate of the finfish infrastructure), adapting a truly *integrated* MTA approach would require some degree of shared infrastructure that would demand appropriate engineering and due consideration of system anchoring, as well as system servicing and operational accessibility for each culture component. In terms of the latter, the finfish component of a proposed *i*-MTA would need unimpeded access to the cages for feeding, grading, net changing (or maintenance), harvesting (with large vessels), etc. The shellfish component, depending upon design of the suspension infrastructure and proximity to the finfish netcages, could (for example) require design modifications for shellstock deployment and retrieval at such a site.

In coastal British Columbia the deepwater commercial shellfish aquaculture candidates for the bivalve component of a Multi-Trophic Aquaculture (MTA) system include oysters, mussels, and scallops. The infrastructure commonly employed in this aquaculture industry sector includes surface longlines (typical for shucked oysters, mussels; see Figure 76-A), subsurface longlines (scallops; see Figure 76-B), and rafts, a relatively new approach that is quickly becoming a standard for oyster (and some mussel) culture, while slowly replacing the use of surface longlines; see Figure 76-C.

Figure 76: Examples of the three primary approaches to suspended shellfish aquaculture in coastal British Columbia. **A.** Surface longline system supporting oyster and/or mussel culture; example shows twin-barrel support for suspended oyster trays used in Lemmens Inlet, west Vancouver Island. **B.** Subsurface longline system supporting scallop nets and or droplines with ear-hung scallops; example from 20-hectare farm in Baynes Sound, east Vancouver Island. **C.** Raft system for tray oyster culture; Baynes Sound, east Vancouver Island.



As noted above, the development of a finfish-shellfish *a*-MTA system would not, generally, be constrained by any particular selection of shellfish culture infrastructure for use at a farm site. Given that *a*-MTA would consider a water quality exclusion zone (WQEZ) around the finfish netcage system so as to minimize (or avoid) the seafood safety risks associated with near-field water quality impacts, this *adjacent* culture approach need not consider the limiting infrastructure conflicts inherent with placing shellfish culture systems in close proximity to the finfish facility. For example, use of shellfish longlines and their associated droplines would not, under typical operational conditions, be at risk of entanglement with the finfish system anchoring system or with the tensioning buoys, billowing nets, or with any of the walkway components when placed outside of the WQEZ. In these cases, any of the shellfish aquaculture approaches could be considered in an integrated finfish-shellfish operation, with the magnitude of the shellfish component constrained only by space available outside of the 'water quality exclusion zone', conflicts with any finfish anchoring gear, or with the biophysical conditions appropriate for the shellfish species and its supporting culture infrastructure.

7.2 Environmental Benefits of a Finfish-Shellfish *Integrated*-MTA System

The design of a true polyculture, or *integrated* Multi-Trophic Aquaculture (MTA) system, takes advantage of the process by which "*one species waste is another's treasure*" (Chopin, 2004). In such an aquaculture design, the system is balanced so as to ensure that the wastes generated from one trophic level are consumed by the next, with the efficiency of the co-culture linkage determined by the proportion of these wastes that are used. In a well-designed MTA system the environmental benefits are realized through the measurable elimination of these waste constituents between the co-cultured species.

In a proposed finfish-shellfish MTA, the reduction in particulate organic wastes generated from the finfish component could be realized through the filtration of water downstream of the farm infrastructure by an established shellfish component. Given that marine finfish aquaculture typically employs an open netcage system, the proportion of the generated wastes consumed by a shellfish component (and hence the direct environmental benefit of this MTA linkage) would be determined by the dispersion pathway of those wastes, the physical characteristics of the waste (e.g., particle size distribution, settling rate), and the population size and structure of the system's shellstock.

This research initiative has indicated that shellfish located downstream of a salmon farm site do ingest farm wastes, as demonstrated by the near-field shellfish samples that revealed elevated levels of contaminant that are associated with these wastes. However, growth rates of the two species employed in the study were not significantly different than that of background, implying that organic waste material released into the natural (or background) seston field does not enhance growth. With respect to contributing to the shellfish growth, as a proportion of this seston, the lack of elevated waste markers (e.g., trace metal constituents of the feed) in all but a few samples (those occurring periodically and in very close proximity to the finfish infrastructure), would suggest that: (i) the contribution of this introduced organic material represents an extremely small proportion of the natural seston field; and/or (ii) the dispersion of these organic wastes is such that the majority of this material is rendered unavailable to the shellfish. Cheshuk et.al.. (2003) drew a similar conclusion in evaluating the culture performance of mussels which were integrated with Atlantic salmon (*Salmo salar*) in Tasmania. In that study, it was noted that salmon farm particulate discharge was not significantly represented within the natural seston, the farm waste did not contribute to an enhanced phytoplankton field, and the integrated shellstock was not grown close enough to the salmon component to result in any measurable increase in such performance.

The natural levels of seston will vary considerably, both temporally (seasonal change) as well as spatially (coastal versus offshore, regionally). In eastern Canada, an evaluation of mussel-finfish MTA (Robinson, 2003) documented low natural seston levels, with a clear finfish organic waste signal measurable above reported background levels. The fact that the introduced waste component contributed significantly to the bioavailable organic matter resulted in a measurable enhancement in shellfish growth at these salmon farms. In contrast, studies of the coastal seston fields in western Canada (Herlinveaux, 1972) indicates that natural levels are comparatively high, and as such may actually mask the organic signal of finfish wastes discharged to coastal waters of British Columbia. Given the low organic waste flux at finfish farms (Chapter 3), the proportion this organic material would represent within naturally high levels of seston may contribute little (if anything) to enhancing shellfish growth or to affecting tissue quality through bioaccumulation of any of the affiliated contaminants.

The natural levels of seston may, in fact, be sufficiently high so as to fully satisfy the nutritional requirements of the shellfish, with these natural food resources rarely becoming limited throughout the life cycle of the species. Depressions in available seston do occur during the winter months, and shellfish positioned in close proximity to a finfish farm may experience enhanced growth if the discharged waste organic material increases in proportion of the available seston within the receiving waters. Slightly higher shellfish growth rates were noted immediately adjacent to the farm sites examined in this study, suggesting a minor contribution to observed growth, although these increases were not statistically significant.

The documented dispersion of organic wastes from the two finfish study sites, as discussed in Chapter 4, and the physical oceanographic processes determining how such dispersion occurs (Chapter 3), suggest that the infrastructure configuration and orientation will contribute significantly to how farm-derived organic wastes may be distributed and potentially utilized by an integrated shellfish component. Size and shape of the net-pens, net-pen spacing, and entire system alignment with respect to the oceanographic and physiographic characteristics of a farm site (e.g., tidal flows, bathymetric and topographic features) will influence the dispersion and dilution of wastes and the potential bioavailability of these wastes to integrated aquaculture species.

These factors, important for minimizing the benthic impacts of finfish farms, can also be used to demonstrate the indirect environmental benefits of integrated finfish-shellfish aquaculture. Assessing the environmental 'foot-print" produced by a finfish aquaculture facility, and comparing it with the environmental impact resulting from the system design/configuration required of an *integrated*-MTA development, will help demonstrate the additional environmental benefits that could be realized through such a development initiative.

The use of mathematical models to predict waste dispersion can provide a quick comparative approach with which to examine the environmental consequences of waste dispersion around a farm system, and for assessing the resulting environmental effects of re-configuring a finfish aquaculture system for integrating a shellfish (or another) trophic-level component within the design of an MTA system. Numerous dispersion models have been developed for assessing benthic effects of salmon aquaculture (e.g., Cromey, 2001; Perez et al. 2002; Perez et al., 2003). In Scotland, the development of DEPOMOD by the Scottish Association of Marine Science, in cooperation with SEPA and Marine Harvest Scotland, was initially intended as a tool for documenting organic waste dispersion and the biological impacts associated with the flux of these organics to the benthic community. Although SEPA currently employs the model for chemotherapeutant 'consents', and does not yet use the model in the context of predicting waste discharge foot-prints, the Canadian salmon farming industry is validating this model for use in objectively defining the spatial extent of organic waste effects and to help develop an appropriate habitat compensation policy for such impacts (Federal Department of Fisheries and Oceans, 2004/5).

In light of this Canadian initiative, the DEPOMOD (version 2.2) model was used to predict the spatial impacts of organic waste flux from the Young Passage study farm, to determine whether measured solids flux (Chapter 4) were accurately reflected in the model predictions, and (if the model performed reasonably) to run simulations that would examine the environmental effects of re-configuring the finfish system (while maintaining comparable finfish production levels) to integrate a commercial level of shellfish production at this site.

The weekly feed deployment record, acquired across the entire production cycle, was used to calculate an average daily feed input to each cage (475 kg/cage/d), and assumed an equal deployment of this quantity for each cage in the system. Following the initial model run, using the actual cage configuration at the site and the feed data for the last production cycle (hind-cast model run), three additional scenarios were considered: (i) a finfish aquaculture system comprised of 14 circular cages established in a standard industry grid at the site, with the additional 2 cages accounting for the reduced 'per-cage' density in such a site; (ii) the 12-cage steel system currently operating at this site re-configured (expanded) to allow paired spacing for an integrated shellfish component; and (iii) an expanded grid of circular cages (14) that would allow a shellfish component to be integrated within the grid. All model settings and inputs, with the exception of cage positions, were kept constant across each of the model runs to allow a direct comparison among the generated benthic impact 'foot-prints'.

Figure 77 (top plot) illustrates the benthic impact foot-print predicted through the DEPOMOD model run for the existing 12-cage salmon farm system. The extent of the organic waste dispersion, shown as depositional contours of organic solids/m^2/day, revealed that the majority of wastes occurred directly beneath the system with localized concentration within the center area. The maximum predicted flux exceeds 24 g solids/m^2/day beneath the system, with declining levels extending towards the perimeter and beyond to approximately 30 meters.

The observed deposition rates at Young Passage (Chapter 4) would suggest that the model very slightly underestimates the dispersion of these wastes, and thus the flux rates at specific distances from the netcage system. Where direct measurement indicated that these wastes were distributed out to approximately 50 meters (>1.0 g/m^2/day within this area), the model predicts that this flux extends out only to approximately 30 meters. Similarly, predicted flux at the edge of the system is approximately 50% lower than that of the measured values, although estimates of seafloor values directly below the cages are within the range assessed at Young Passage in this study (beneath cage = 17.11 g solids/m^2/day; cage edge = 13.89 g solids/m^2/day).

Although there are quantifiable differences between observed and predicted flux rates of organic wastes (solids) at the Young Passage site, the dispersion pattern appears reasonable and use of the model for subsequent evaluations of environmental effects given changes in system configuration should provide sufficient data for gross comparisons if not for 100% reliable data on waste flux. The differences noted could be attributed to inherent limitations in this particular model, including the fact that data from a fixed-point current meter mooring are used to describe the hydrodynamics across the entire model domain, and that drift in the cages over a tidal cycle are not accounted for within the model itself. In addition, the field data may contribute to this difference by reflecting higher organic waste fluxes due to the fact that the employed sediment canisters 'trapped' the waste during settling and as such could not allow for any re-suspension and re-distribution of these wastes once they were retained within the collection system.

Figure 77 (bottom plot) illustrates the spatial impact 'foot-print' at Young Passage when a circular netcage system grid is used rather than the 12-cage steel system described above. Given the 10-15 meter separation of cages within a circular grid system, the difference in predicted organic waste flux is considerably less than that reported for the consolidated steel cage system. The maximum waste flux predicted for this system is 12 g solids/m^2/day. The pattern of dispersion is similar to that predicted for the 12-cage steel system, with the waste contours centered on the long axis of the grid and extending only a short distance beyond the perimeter of the system.

Figure 77: Predicted organic waste dispersion at Young Passage farm site using DEPOMOD (vers.2.2) **A.** steel net-pen system as currently operated at farm site; **B.** circular cage system grid with same level of production and feeding rate (per cage). Results as *g solids /m.sq./day.*

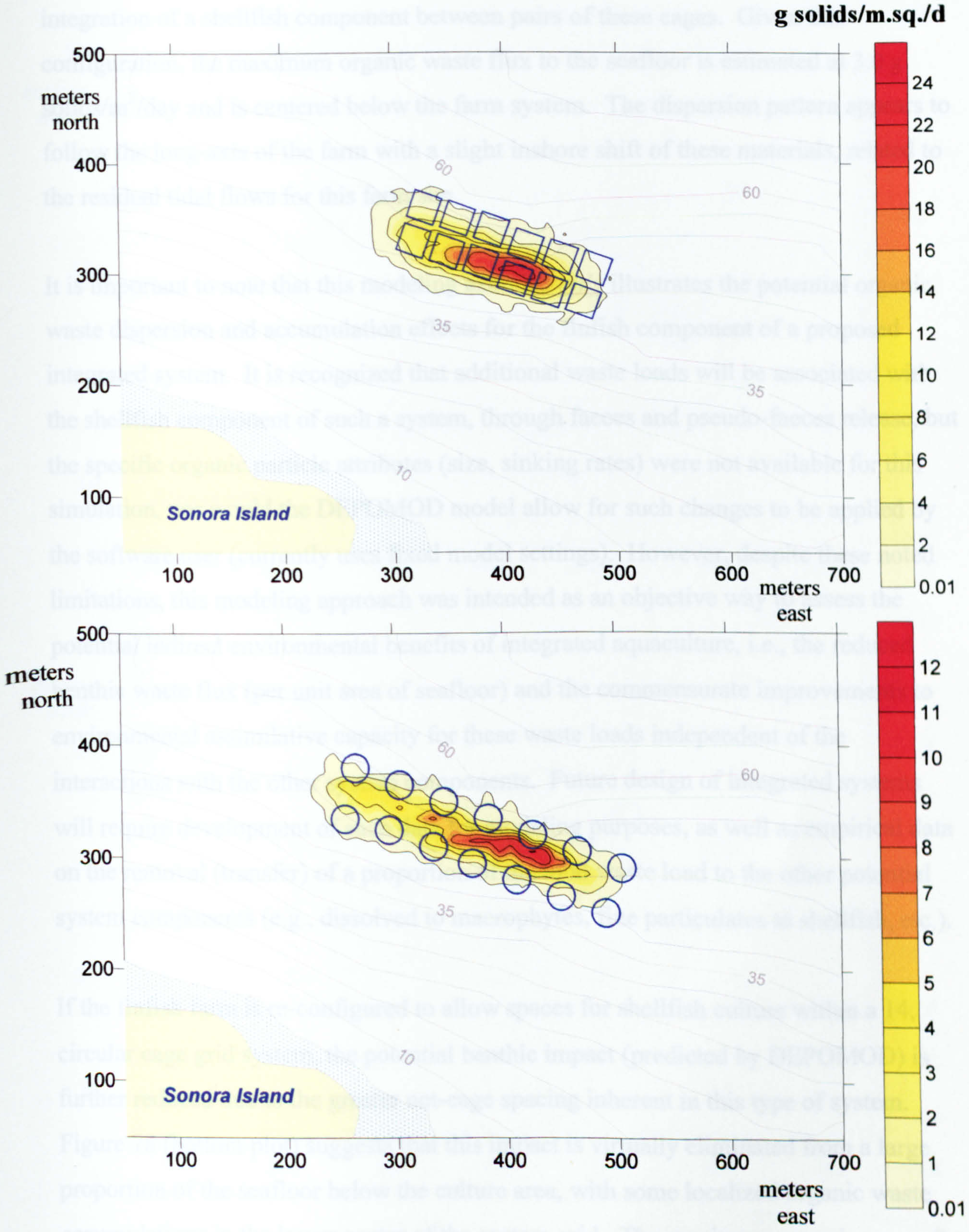
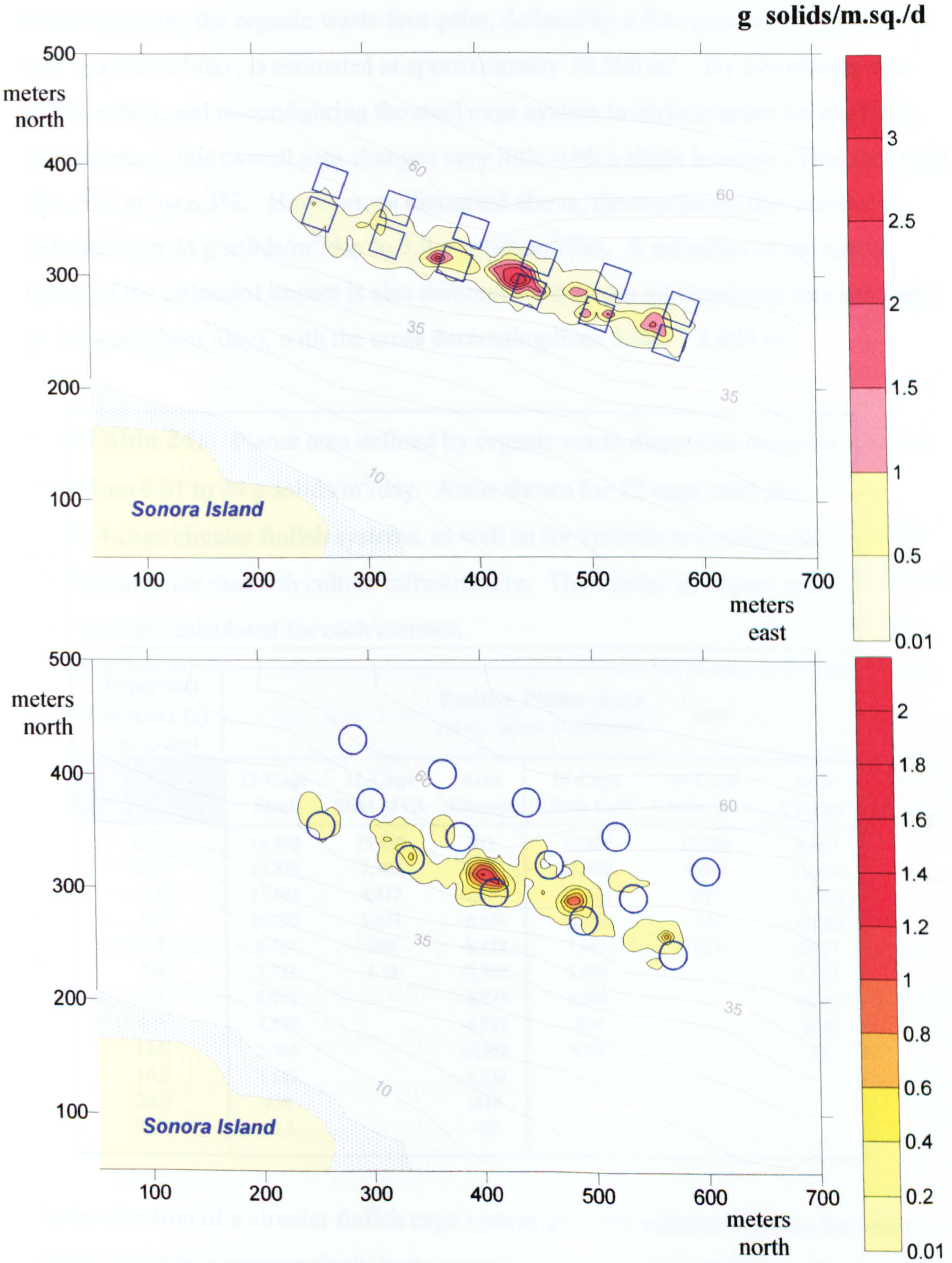


Figure 78 summarizes the DEPOMOD model predictions of waste material dispersion and benthic accumulation at the Young Passage farm site with net-cage systems re-configured to integrate a shellfish aquaculture component. The upper figure illustrates the benthic impact foot-print from the steel cage system re-configured to allow integration of a shellfish component between pairs of these cages. Given this configuration, the maximum organic waste flux to the seafloor is estimated at 3.0 g solids/m^2/day and is centered below the farm system. The dispersion pattern appears to follow the long-axis of the farm with a slight inshore shift of these materials, related to the residual tidal flows for this farm site.

It is important to note that this modeling exercise *only* illustrates the potential organic waste dispersion and accumulation effects for the finfish component of a proposed integrated system. It is recognized that additional waste loads will be associated with the shellfish component of such a system, through faeces and pseudo-faeces release, but the specific organic particle attributes (size, sinking rates) were not available for this simulation, nor would the DEPOMOD model allow for such changes to be applied by the software user (currently uses fixed model settings). However, despite these noted limitations, this modeling approach was intended as an objective way to assess the potential indirect environmental benefits of integrated aquaculture, i.e., the reduced benthic waste flux (per unit area of seafloor) and the commensurate improvements to environmental assimilative capacity for these waste loads independent of the interactions with the other system components. Future design of integrated systems will require development of such data for modeling purposes, as well as empirical data on the removal (transfer) of a proportion of the total waste load to the other potential system components (e.g., dissolved to macrophytes, fine particulates to shellfish, etc.).

If the finfish farm is re-configured to allow spaces for shellfish culture within a 14, circular cage grid system, the potential benthic impact (predicted by DEPOMOD) is further reduced due to the greater net-cage spacing inherent in this type of system. Figure 78 (bottom plot) suggests that this impact is virtually eliminated from a large proportion of the seafloor below the culture area, with some localized organic waste accumulations in the lower center of the system grid. The maximum organic waste flux under this scenario is predicted at 2.0 g solids/m.sq./day.

Figure 78: Predicted organic waste dispersion at Young Passage farm site using DEPOMOD (vers.2.2) **A.** steel net-pen system re-configured for finfish-shellfish *i*-MTA site; **B.** circular cage system grid with same level of production and feeding rate (per cage); expanded grid for *i*-MTA system. Results as *g solids /m.sq./day*.



To quantitatively compare the estimated environmental benefits illustrated in the above four production scenarios, Table 24 summarizes the spatial extent of the waste material concentrations over the seafloor, showing the planar area covered by each of the accumulation contours. In the case of the steel 12-cage system that currently operates at this farm site, the organic waste foot-print, defined by a flux greater than or equal to 0.01 g solids/m^2/day, is estimated at approximately 14,500 m^2. By introducing an *i*-MTA system, and re-configuring the steel cage system to include space for shellfish infrastructure, this overall area changes very little with a slight increase of the foot-print size, 912 m^2 or 6.3%. However, as discussed above, the maximum flux contour is reduced from 24 g solids/m^2/day to 3.0 g solids/m^2/day. A reduction in the spatial extent of the estimated impact is also demonstrated across all remaining flux contours (< 3.0 g solids/m^2/day), with the areas decreasing from 5,000 – 8,600 m^2.

Table 24: Planar area defined by organic waste dispersion contours from 0.01 to 24 g solids/m^2/day. Areas shown for 12-cage steel and 14-cage circular finfish systems, as well as for systems re-configured to integrate shellfish culture infrastructure. The change in impact areas are also calculated for each contour.

Dispersion Contours (z) *(g/m.sq./day)*	**Positive Planar Area** *(m.sq.; above z=contour)*					
	12-Cage Steel	**12-Cage Steel MTA**	**Area Change**	**14-Cage Circle Grid**	**14-Cage Circle MTA**	**Area Change**
0.01	14,498	15,410	912	17,255	12,288	-4,967
0.25	12,322	7,309	-5,013	13,420	2,014	-11,406
0.5	11,481	4,412	-7,069	12,015	957	-11,058
1.0	10,292	1,671	-8,621	10,219	139	-10,080
2.0	8,727	308	-8,419	7,683	16.1	-7,667
3.0	7,751	1.18	-7,750	5,695		-5,695
4.0	6,975		-6,975	4,044		-4,044
8.0	4,529		-4,529	834		-834
12.0	2,562		-2,562	9.99		-10
16.0	1,518		-1,518			
20.0	618		-618			
24.0	22.3		-22			

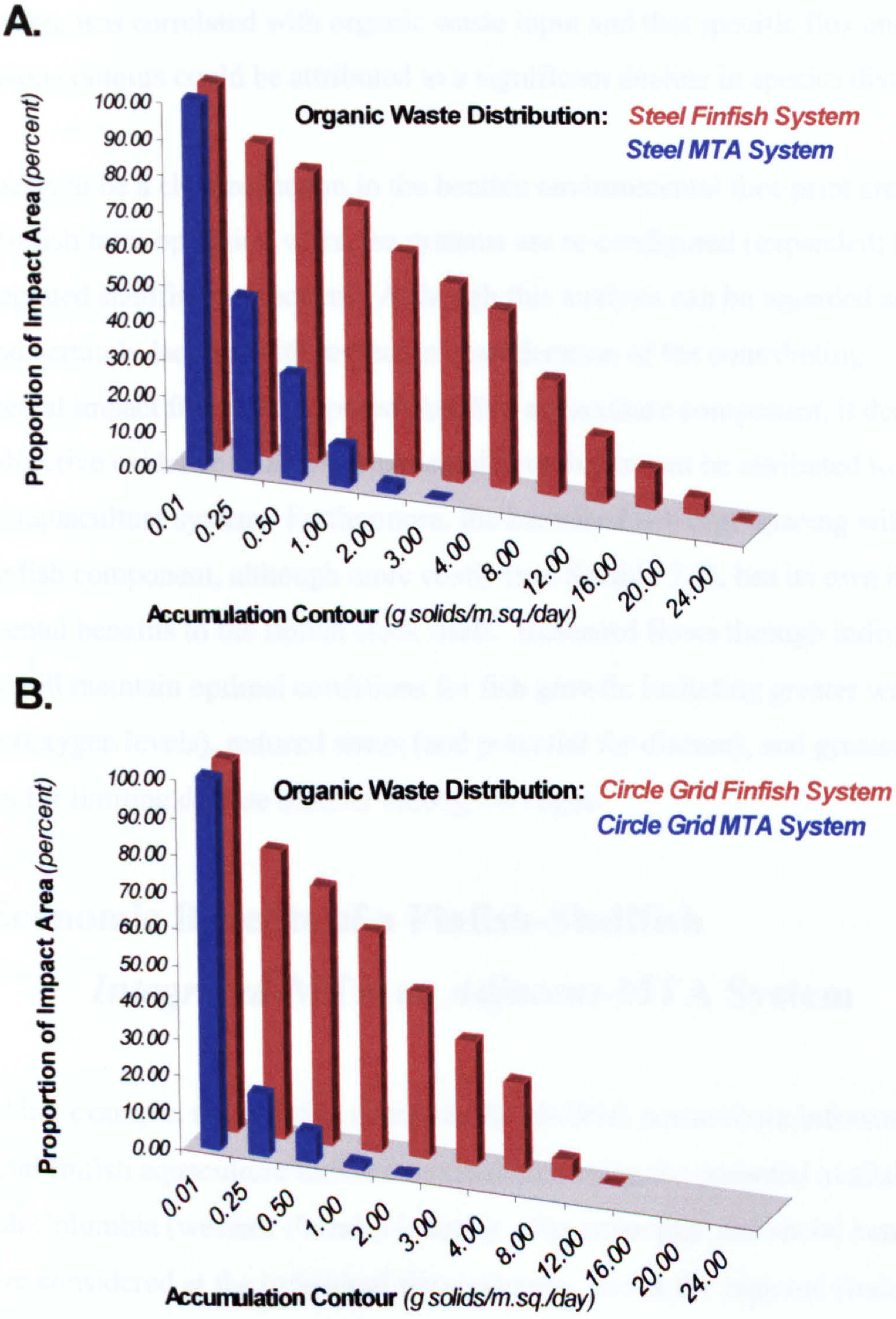
In the situation of a circular finfish cage system grid, the inherent spacing between cages produces a comparatively large organic waste foot-print (17,255 m^2), with much of the waste material overlapping to form a continuous foot-print at lower levels of flux. Re-configuration of this grid to a 14-cage circular MTA system results in a significant drop in overall waste impact foot-print (12,288 m2) which represents a 4,967 m^2, or 28.8%, reduction in size. Furthermore, the MTA system increases the between-cage distance to a point where impacts become largely independent, rather than cumulative, beneath the system.

Figure 79 illustrates the reduction in organic waste flux to the benthic environment by comparing the change in benthic impact contours predicted by DEPOMOD simulations for the steel and circular grid finfish culture systems upon reconfiguration to *i*-MTA systems. In this summary, contour data (planar areas) acquired from Table 24, are represented as proportions of the entire environmental foot-print as defined by the 0.01 g solid/m^2/day contour.

Figure 79A compares the steel cage system under the two configurations, the red bars showing the proportion of the entire impact area (~15,000 m^2) affected by waste accumulations (each of the 12 waste flux contours) under the traditional cage system configuration (finfish component only), while the blue bars illustrate the distribution of the waste when the steel system has been re-configured for use as an *i*-MTA system. The distribution shows that considerable material is concentrated directly beneath the system, with organic waste flux between 3 and 24 g solids/m^2/day under the finfish system. With the hypothetical *i*-MTA system, waste flux is generally less than 3.0 g solids/m^2/day across the entire impact area, with the majority being deposited at a rate of $\leq$ 0.5 g solids/m^2/day.

Figure 79B presents this comparison for the circular grid system. Again, the overall environmental effects are dramatically reduced once the cage system is re-configured to include an integrated component. The predicted dispersion, and resulting spatial accumulation of organic waste, is significantly reduced across the site. Spatial extent of the individual impact contours (above 0.01 g solids/m2/day) also decline significantly in response to this system configuration (loss of 5,000 – 11,500 m^2).

Figure 79: Distribution of organic waste material across accumulation contours (g solids/m^2/day) as predicted by DEPOMOD simulations; data expressed as proportion of 0.01 g solids/m^2/day contour size (percent). **A.** Comparison of 12-cage steel finfish system with proposed *i*-MTA system configuration, and with **B.** 14-cage circular grid finfish system.



The benthic effects (physical-chemical, biological) will also be measurably reduced as a consequence of facilitating a well-dispersed organic waste. Studies have shown that an organic flux greater than 2.0 g solids/m^2/day will be reflected in a change to sediment chemistry (e.g., increased free sulfides) which, in turn, corresponds to the degradation of the resident benthos (Hargrave, 1997; Brooks, 2001). Cromey (2001) revealed that the health of the benthos community, measured through the Infaunal Trophic Index, was correlated with organic waste input and that specific flux and waste accumulation contours could be attributed to a significant decline in species diversity.

There appears to be a clear reduction in the benthic environmental foot-print created by a typical finfish farm operation when the systems are re-configured (expanded) to allow for an integrated shellfish component. Although this analysis can be regarded as coarse, and certainly lacking with respect to consideration of the contributing environmental impact from the proposed shellfish aquaculture component, it does provide objective evidence of an environmental benefit that can be attributed to an integrated aquaculture system. Furthermore, the increased net-cage spacing within the *i*-MTA finfish component, although more costly (see Section 7.3), has its own inherent environmental benefits to the finfish stock itself. Increased flows through individual net-cages will maintain optimal conditions for fish growth, including greater water exchange (oxygen levels), reduced stress (and potential for disease), and greater separation for limiting disease transfer among the cages.

7.3 Economic Benefits of a Finfish-Shellfish *Integrated*-MTA or *Adjacent*-MTA System

As a working example, the addition of suspended shellfish aquaculture infrastructure to commercial finfish aquaculture facilities is examined using the potential available to the British Columbia (western Canada) industry. The economic and social benefits of i-MTA are considered at the individual site, company, and at the regional (industry) levels. The potential realized through evaluation of the western Canada scenario is further used to provide insight into MTA development opportunities for other temperate latitude countries.

7.3.1 Corporate Opportunities

In addition to the environmental factors that will determine whether or not *i*-MTA is technically feasible at a specific farm site, the integration of a shellfish component to an existing finfish operation must, from a corporate perspective, consider the economic benefits of pursuing such a development. The scale of operation (level of production), capital and operating costs, market stability, and potential return on investment are all critical business criteria upon which the risks of such a venture must be weighed.

To explore the economic potential and to evaluate the corporate risks associated with the development of a finfish-shellfish *i*-MTA system, the Young Passage study site has been used as a hypothetical example of such a development. The existing cage system was modified to allow for the integration of shellfish rafts, supporting mussels, oysters, and/or scallops. For this evaluation, the latter two shellfish taxa were used.

To integrate the shellfish component, the Young Passage finfish system is re-configured (expanded) to maintain current production levels for salmon (2,500 MT), employing six pairs of steel 30 x 30 meter net-cages, while creating space equivalent to 10 of these steel cages for the shellfish component (total shellfish area = 9,000 m^2); see Figure 80. The entire, integrated system would occupy a floating (surface) area of 348 x 67 meters (2.33 hectares). The center walkway would extend the length of the new system, as would the feed distribution lines for the finish cages, each representing an additional capital cost for the proposed *i*-MTA system (over that of the existing finfish infrastructure).

Each of the ten shellfish component 'holes' (30 x 30 meters) could, conservatively, support four shellfish rafts, each 8 x 8 meters (the present industry standard in British Columbia). This configuration would allow a 4-meter separation between the rafts, as well as a 5-meter clearance from the adjacent finfish net-cages (Figure 81). A series of smaller rafts, rather than a single large structure, would ensure adequate water flow among the shellstock, eliminating the potential impacts to growth and survival on individuals at the center of such a large concentration of filter feeders; extremely high densities of either *i*-MTA component could have an unsustainable impact on available oxygen levels, and thus on the health of the entire integrated system. The separation from the adjacent net-cages would also reduce (or eliminate) potential physical and operational conflicts with the finfish aquaculture component of the *i*-MTA.

Design and installation of a shellfish raft system within an *i*-MTA system could take advantage of the available (adjacent) steel net-cage system components. Mooring of the raft system within the steel system could rely primarily on the adjacent finfish system (3-sides), and thereby require only a single offshore anchor for the 4-raft system (Figure 81). Maintaining rigidity in the 4-raft configuration could also allow for design of a ramp/walkway to access the shellfish system via the main system walkway. This feature could reduce (or eliminate) the need for independent access of the shellfish system by boat, and thus result in significant savings in operational costs for the shellfish component of the *i*-MTA facility. If this design proved useful, the steel outer walkway (typical of a finfish component – without the net) could be continued around the entire 4-raft system, applying further structural integrity to the system and protection from normal component wear and that associated with extreme weather events.

A single raft can support considerable shellfish product, and use of these deepwater production systems have eliminated the need for the extensive growing areas necessitated when employing the traditional surface or subsurface longlines. In terms of oyster production, a typical raft is configured with trays to support stacks of single oysters. Seed are grown to market size (7-9 cm), removed from the trays and transferred to a beach lease to allow the shell to harden before processing.

For scallop culture, although not typically grown on rafts, these animals could be cultured in nets or ear-hung on individual droplines suspended from the rafts rather than from the usual longline culture system. For the purpose of this hypothetical configuration and assessment, nets are used to grow seed scallops to 3-cm juveniles, and it is assumed that these would then be ear-hung and grown out to market size (12 cm shell diameter) on droplines deployed directly from the *i*-MTA raft system.

Figure 80: Hypothetical layout of finfish-shellfish integrated-MTA system at Young Passage. The *i*-MTA is configured using an elongated steel net-cage system with ten clusters of 4 shellfish rafts interspersed within the rectangular array. Center walkway of steel cage system is indicated, as is the location of feed barge (and on-site accommodation) facility.

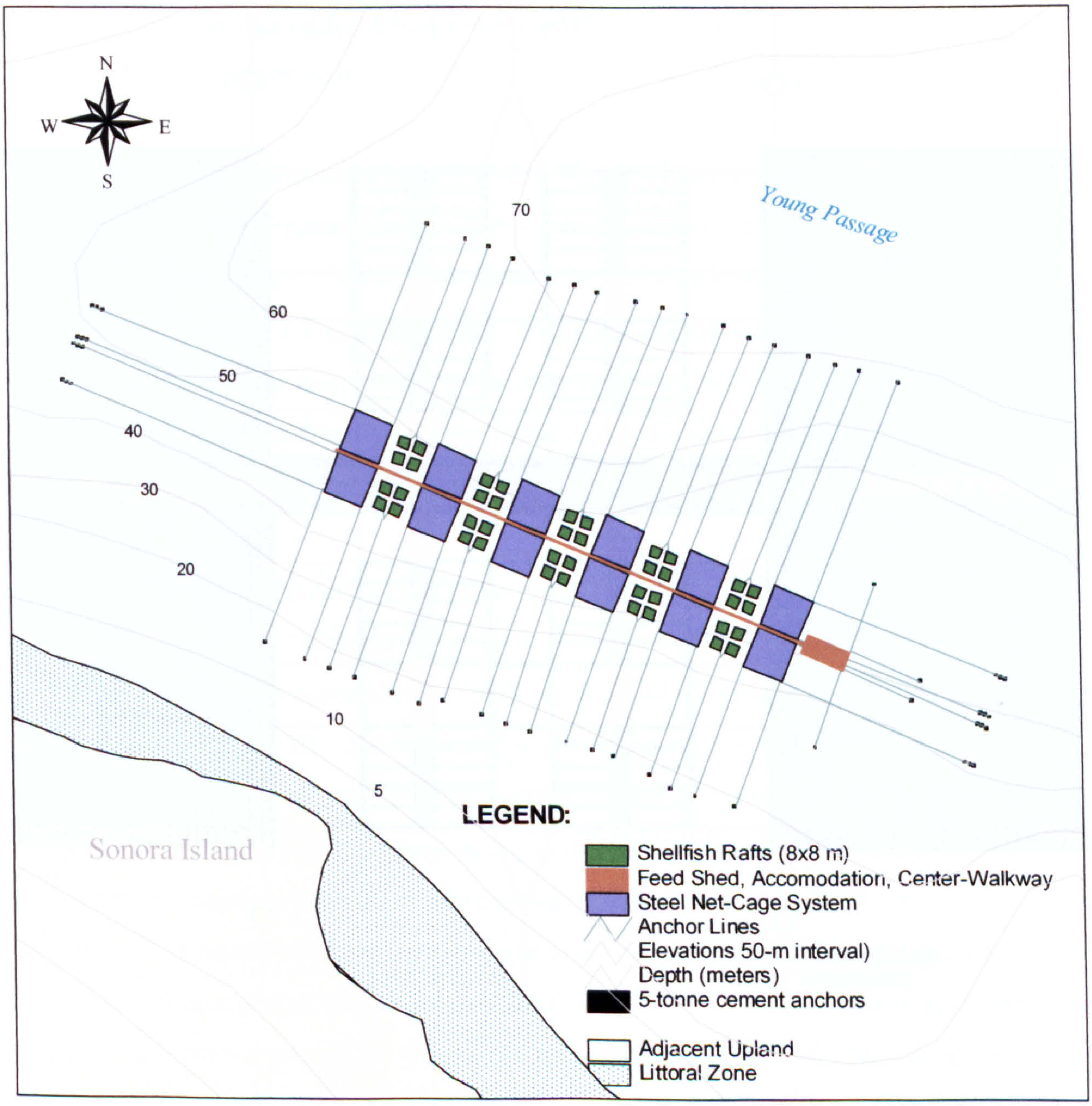
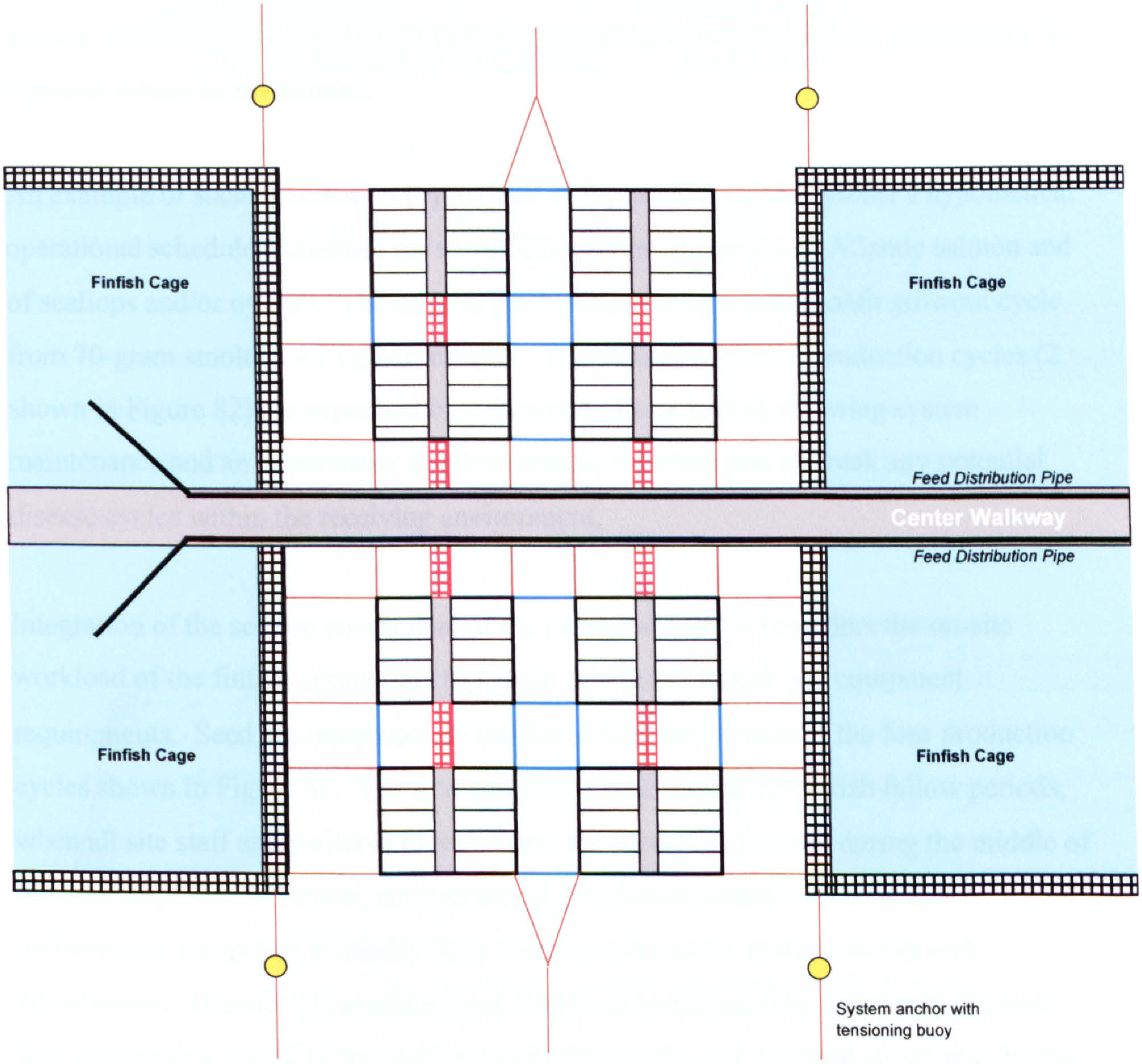


Figure 81: Possible design of shellfish aquaculture component of a finfish-shellfish *integrated*-MTA system. A 4-raft system replaces single 30x30 meter net-cages. Rafts are stiff-legged (blue lines) to maintain relative positions. Mooring system uses the adjacent net-cages (dark red lines) as well as a single off-system anchor. Red 'walkway' could be engineered to access rafts from center system walkway, eliminating need for independent (boat) access to shellfish component for seeding, harvesting, and other servicing requirements.

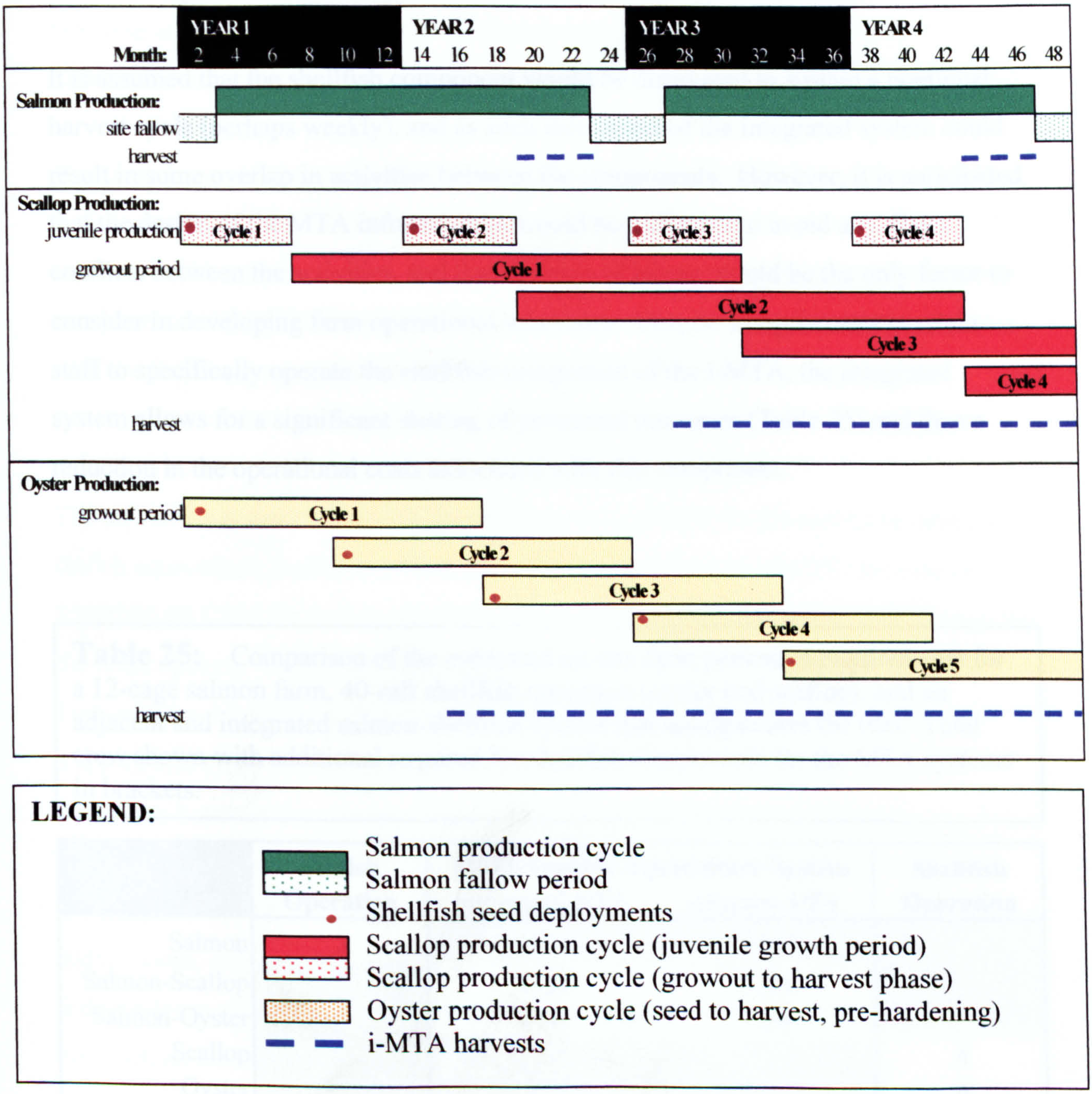


The *i*-MTA system could be operated in a variety of ways, depending upon the species proposed for co-culture, the planned level of production, and the market demand (product size, quantity, delivery frequency). In assessing the economic potential of finfish-shellfish aquaculture, the efficiencies of integrating the two culture components must also be considered in the context of operational planning and implementation. Eliminating equipment redundancies, optimizing use of site personnel, and joint planning of activities/schedules across the MTA components, will all contribute to process efficiencies that will result in lower operating expenditures and, presumably, in a greater return on investment.

An example of such efficiencies is provided in Figure 82, which presents a hypothetical operational schedule (timeline) for an *i*-MTA system comprised of Atlantic salmon and of scallops and/or oysters. The salmon production assumes a 20-month growout cycle from 70-gram smolt to 4.5 kg harvest fish. The proposed salmon production cycles (2 shown in Figure 82) are separated by a 4-month fallow period, allowing system maintenance and an opportunity to allow benthic recovery and to break any potential disease cycles within the receiving environment.

Integration of the scallop component of the proposed *i*-MTA considers the on-site workload of the finfish component to reduce conflict in labour and equipment requirements. Seed are introduced to the site at the start of each of the four production cycles shown in Figure 82. The first and third occur during the finfish fallow periods, when all site staff are available to assist, and the second and fourth during the middle of the salmon production cycle, corresponding to a period within which finfish maintenance comprises primarily daily feeding and routine system checks and adjustments. Transfer of juveniles from nets to ear-hanging rafts represents the most time-consuming aspect of the scallop production cycle, and seasonal crews may be the most cost effective approach for dealing with this stage of each cycle. Nevertheless, the timing for this activity would also occur early in the finfish production cycle and is planned, in this scenario, to avoid those periods of increased activity in the salmon production cycle.

Figure 82: Production cycles for proposed *i*-MTA components over 4 years. Atlantic salmon (*Salmo salar*) grown to 4.5 kg, and each production cycle is followed by a 4-month fallow period. Scallop component comprises juvenile rearing as well as growout phase for 4 production cycles. Oysters seed deployed twice annually to produce continual product after first year a total of 5 cycles are shown for this example of an integrated production schedule.



For an oyster component of an *i*-MTA, seed introduction could also be timed to avoid conflict with the finfish production activities. With a shorter growth period than scallops, oyster seed could be introduced to the system at three times over the finfish cycle. The first would occur within the fallow period, the subsequent two after smolt entry and in the later portion of the finfish cycle (prior to harvest) to take advantage of available farm staff for the seed deployment process.

It is assumed that the shellfish component would be developed to sustain a continual harvest cycle (perhaps weekly), and as such this aspect of the integrated system could result in some overlap in activities between the components. However, it is anticipated that the design of the MTA infrastructure would be sufficient to avoid any direct conflicts between the activities, and that division of labour would be the only factor to consider in developing farm operational schedules. Despite a requirement to retain staff to specifically operate the shellfish component of the *i*-MTA, the integrated system allows for a significant sharing of personnel resources (Table 25) and thus a reduction in the operational costs associated with this component.

Table 25: Comparison of the estimated on-site farm personnel requirements for a 12-cage salmon farm, 40-raft shellfish operation (oyster and scallop), and an adjacent and integrated salmon-shellfish system that amalgamates the two. Total crew shown with additional required for shellfish components for the MTA systems in brackets.

	Finfish Operation	**Multi-Trophic Aquaculture System**		**Shellfish Operation**
		integrated-MTA	*adjacent-MTA*	
Salmon	2			
Salmon-Scallop		4 (2)	5 (3)	
Salmon-Oyster		6 (4)	7 (5)	
Scallop				4
Oyster				6

The estimates provided in Table 25 assumes that special crews are employed to ear-hang scallops, deliver seed, and to transport product post-harvest, while the on-site crews would be responsible for all other aspects of the shellfish component. The opportunity for sharing labour resources between the *i*-MTA components results in an estimated 50% reduction in staffing requirements for an added scallop component, and a 33% reduction for an oyster component. It is suggested that *adjacent*-MTA, although benefiting from the shared staffing concept, would be less efficient than that of the *integrated*-MTA due to the physical separation of the system components and the inherent technical issues associated with operating an independent raft system(s). Given an *a*-MTA approach, labour efficiencies are estimated to increase from 8% to 25% for oyster and scallop components, respectively.

The capital expenditures associated with the integration of shellfish within an existing finfish operation are presented in Table 26. A comparison of *integrated*-MTA and *adjacent*-MTA (similar costs to that of an independent shellfish operation) indicate the components of the MTA system where costs are saved as a consequence of integration with the finfish component, and where additional costs must be incurred in the development of the supporting infrastructure for such a system.

The capital costs associated with the addition of a shellfish component to an existing finfish aquaculture facility (*i*-MTA) are estimated at $502,145.00 CDN for scallops and $589,095.00 CDN for oysters, the difference attributed primarily to the cost of trays for the latter species. For introducing an adjacent shellfish component (*a*-MTA), in the form of a 4 x 10 raft grid, the costs are slightly greater, with scallops requiring $522,125.00 CDN and oysters $609,075.00 CDN of capital investment. The cost difference between the two approaches, although very minor (approximately $20,000.00 CDN or < 3.3%), is associated with the cost of the finfish system modification for the *integrated*-MTA and the additional anchoring requirements for the independent, *adjacent*-MTA system. The a-MTA system, given its size and independence of the finfish infrastructure, would also require an additional workboat to run such an extensive operation.

Table 26: Estimated capital costs associated with the shellfish and finfish aquaculture components of an *integrated* and an *adjacent* MTA system. Only the costs of infrastructure required in addition to the base 12-cage steel finfish system are provided for comparative purposes. The adjacent-MTA system is assumed to comprise a 4 x 10 (40) raft system in an anchor grid.

MTA System Component	**Infrastructure Item(s)**	**Multi-Trophic Aquaculture System**	
		integrated-MTA	*adjacent-MTA*
Finfish			
	a. center walkway, 2-meter wide steel, five 33-meter lengths (165 meters total)	$85,000.00	$0.00
	b. ten system anchor lines, including chain, 5-MT cement anchor blocks, and 150 meters of polysteel rope (2.5")	$10,000.00	$0.00
	c. 930 meters of PVC feed distribution line	$1,395.00	$0.00
Shellfish			
	a. 40 shellfish rafts (8 x 8 meters)	$180,000.00	$180,000.00
	b. Oysters: trays (8 trays/dropline x 88 droplines/raft x 40 rafts in system) = 28,160 trays	$211,200.00	$211,200.00
	c. Scallops: nets for seed-juvenile production; 15 nets/dropline x 192 droplines x 2 rafts	$10,000.00	$10,000.00
	d. Scallops: droplines for ear-hanging; 225/raft @ 15m = 135km of 1/4" rope	$74,250.00	$74,250.00
	e. on-site ear-hanging system (two)	$40,000.00	$40,000.00
	f. dedicated work boat(s)	$90,000.00	$180,000.00
	g. Anchoring system: i-MTA with single offsystem anchor line and mooring to finfish system (1800m rope + 10 anchors); a-MTA comprised of 4 x10 (40-raft) grid system (5325m rope + 45 anchors)	$11,500.00	$37,875.00
	Total Estimated Capital Additions:		
	Scallops:	**$502,145.00**	**$522,125.00** CDN
	Oysters:	**$589,095.00**	**$609,075.00** CDN

Although a crude estimate, given possible changes in production levels, combinations of shellfish, and farm-gate pricing, Table 27 provides an evaluation of the potential revenues that could be generated from the shellfish component of an integrated-MTA. This assessment assumes that the MTA site is operated independent of other corporate sites, and as such relies on a continuous harvest (revenue) stream generated through the culture of multiple year-class shellstock. In the case of scallops, three year-classes would be maintained at the site while oysters would require two. The total gross revenue generated from a scallop MTA component is estimated at $546,000.00 CDN, while oyster production would gross in the order of $528,000 CDN annually.

While the gross revenues generated through an independent shellfish operation, an *adjacent*-MTA shellfish component, or from an *integrated*-MTA shellfish component would all be the same (given same facility size, and similar stocking/operational approaches), the economic benefits of *i*-MTA are realized through capital cost savings (illustrated above) and by a reduction in operating costs (labour requirements being one of these factors; discussed previously). A summary of these savings, and other operational efficiencies (and thus financial benefits) to be gained through implementation of a MTA system, are presented in Table 28.

The annual operating costs of a shellfish MTA component reveals a substantial saving in costs over that of operating an independent shellfish facility of the same size (production and infrastructure). For an *integrated*-MTA system the operational costs would likely be 35% less, while that of an *adjacent*-MTA system are estimated to save approximately 20% of the typical costs.

Table 27: Estimated site production for shellfish component of a proposed i-MTA. Assessment assumes that the site supports multiple year-class shellstock to satisfy continual harvest requirements. Conservative stocking data and farm-gate prices are used to assess economic potential of these culture approaches.

Raft Infrastructure		Steel Cage i-MTA System		
Finfish System (# of cages)		12		
Finfish 'Holes': *integrated* shellfish component		10		
Number of Rafts/Hole		4		
Total Available Rafts:			**40**	*rafts*
Oysters:	tray stacks/raft (8 by 11 rows)	88		
	trays per stack:	8	**704**	*trays/raft*
Scallops:	seed droplines/raft (16 x 12 rows)	192		
	nets/dropline	15	**2880**	*nets/raft*
	growout droplines/raft (15x15)	225	**225**	*drops/raft*

Scallop Production Potential

Assumes site is multi-year class (3) to provide continual production. One raft is retained for ongoing seed-juvenile production and the remaining 39 rafts are used for ongrowing (ear-hung scallops).

Annual Site Harvest:	13	rafts
	50400	scallops/raft
	655200	scallops produced
	99273	kg. of scallops
	$5.50	farm-gate price/kg. (CDN)
	$546,000.00	**Total Gross**

Oyster Production Potential

Assumes site is multi-year class (2) to provide continual production. 30 rafts are used for each cycle, with adults moved to adjacent beach for hardening prior to harvest.

Annual Site Harvest:	30	rafts
	84480	oysters/raft
	2534400	oysters produced
	211200	dozen
	$2.50	farm-gate price/dozen (CDN)
	$528,000.00	**Total Gross**

Table 28: Estimated operational costs of a 40-raft scallop operation, comparing typical (independent) system with that of a proposed *integrated*-MTA and *adjacent*-MTA culture system. Similar proportions are anticipated for other species of shellfish cultured with this infrastructure. Costs are considered crude, and are intended for comparative purposes only.

Operational Aspects for a Scallop Operation	**Shellfish Aquaculture System**		
	i-MTA	*a-MTA*	*independent*
Labour Requirements:			
full-time (2 shifts)	$144,000.00	$216,000.00	$288,000.00
seasonal (3 months)	$24,000.00	$24,000.00	$24,000.00
Accomodation/Living:			
$25/pers/day inclusive	$13,687.50	$20,531.25	$36,500.00
Seed Purchase/Delivery:	$45,000.00	$45,000.00	$50,000.00
Harvest Transport:			
weekly harvest with live transport via boat and then truck (will vary by site)	$117,000.00	$117,000.00	$156,000.00
Boat Maintenance/Operation:	$12,000.00	$24,000.00	$24,000.00
Capital Equipment Depreciation:	$50,215.00	$52,213.00	$52,213.00
Misc. Expenses:			
(e.g., insurance, license fees, contingency, etc.	$25,000.00	$25,000.00	$25,000.00
Estimated Total Annual:	**$430,902.50**	**$523,744.25**	**$655,713.00**
Percent of Independent:	***65.7***	***79.9***	***100.0***

NOTES:

- Capital equipment depreciation assumes linear model over 10 years to a value of 0.
- Seed costs are the same for each approach, although delivery could be made in conjunction with finfish farm operational requirements.
- Salaries are set at $36,000 per annum

The example described above pertains to the culture of scallops, although it is anticipated that similar such savings would be realized for an MTA oyster operation or for any other shellfish species cultured using such an approach. These operational expenditures will, of course, vary from site to site and will be affected by a variety of factors. In remote coastal areas operational efficiencies become critical in determining the economic viability of a proposed shellfish aquaculture facility, and the development of an MTA system provides the opportunity to capitalize on the infrastructure and operational activities/schedules available through the other culture components (e.g., finfish). In particular, transportation costs (e.g., for crew, supplies, seed, harvest product) represents a significant, and usually limiting factor for developing shellfish in remote regions.

The economic benefits of MTA, in terms of shellfish production, can be seen when comparing the margins (net revenues) generated as a result of the various culture approaches. Table 29 suggests that the culture of shellfish on this scale, in a remote area that would require such operational logistics, would not (given this crude assessment) be economically feasible. The development of shellfish culture independently would require significant capital investment and operational costs, the magnitude of which would result in a annual loss (estimated here at over 20%). These values could be mitigated through reduction in staffing (shifts) and other factors, but the net revenues would still be marginal and would leave little for operational contingency.

The development of shellfish culture within the framework of a Multi-Trophic Aquaculture system appears to mitigate many of the capital and operational risks of developing shellfish culture (in remote coastal areas) independently. Where *adjacent*-MTA takes advantage of primarily the operational efficiencies of the co-culture with finfish, and suggests a net revenue return of 0.8% and 4.1% for oysters and scallops respectively, the development of an *integrated*-MTA improves those returns (18-21%) by optimizing the use of equipment resources and operational efficiencies (Table 29).

Table 29: Estimated Net Revenues (percent of total gross revenues minus operating costs in $CDN) generated by shellfish facility operated as an *integrated* and an *adjacent*-MTA component, as well as an independent operation. Example assumes similar operating costs for scallops and oysters, and proposes a large operation within a remote area (requiring on-site supporting infrastructure).

Scallop Production Potential	**Shellfish Aquaculture System**		
	i-MTA	*a-MTA*	*independent*
Total Annual Revenue	$546,000.00	$546,000.00	$546,000.00
Estimated Operational Costs	$430,902.50	$523,744.25	$655,713.00
Profit Margin (%):	**21.1**	**4.1**	**-20.1**
Oyster Production Potential			
Total Annual Revenue	$528,000.00	$528,000.00	$528,000.00
Estimated Operational Costs	$430,902.50	$523,744.25	$655,713.00
Profit Margin (%):	**18.4**	**0.8**	**-24.2**

The corporate decision to pursue the development of an integrated finfish-shellfish operation would not be based solely on a single site assessment as exemplified above. Although the MTA scenario would likely be tested (at a commercial level of production) at one farm site, the corporate evaluation would also require scrutiny of the economic potential to be realized on a company basis.

As finfish companies typically operate multiple production sites, the capability (discussed previously) to produce shellfish across these sites becomes an important consideration in this economic assessment. In the British Columbia salmon farming region, companies generally operate between 9 and 24 farm sites. Upon review of the site characteristics of the sites within this region, a conservative estimate of 45% can be used to delimit the number of sites with the potential to support a shellfish aquaculture component.

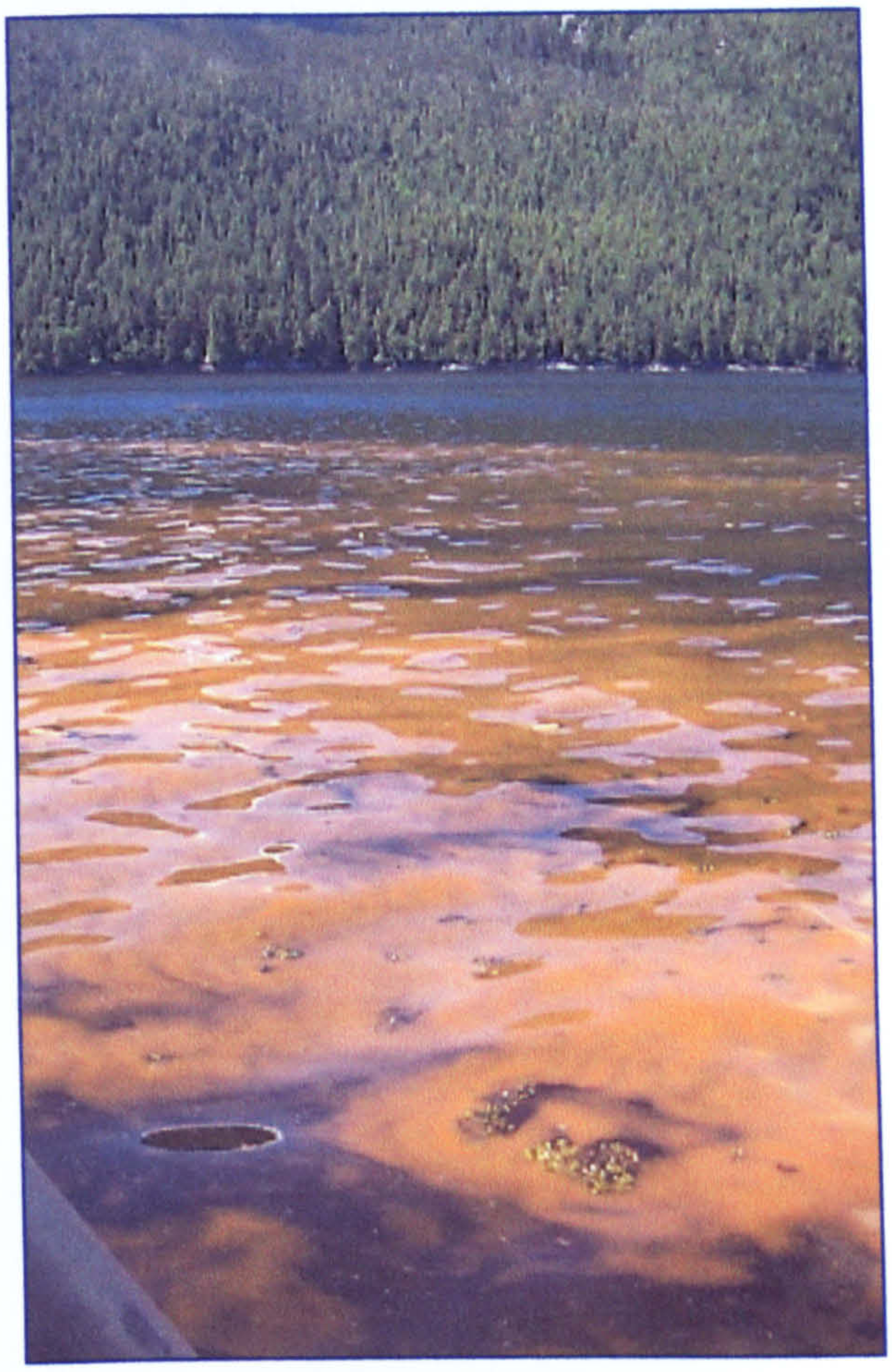
Assuming that an average company maintains 16 salmon farm sites and 7 of these sites could support an MTA system, then the potential net annual revenue (farm-gate profit) generated from such a development would approximate $750,000.00 (Table 30). These are, again, considered conservative estimates given that further economies of scale could be realized through the operation of multiple sites (particularly within a small geographic area). For example, sites could stock a single year-class of shellstock, thereby reducing site-specific workload and allowing for further sharing of farm staff among sites. In addition, all transportation requirements could be scheduled among sites, thereby avoiding any duplication of effort and generating further cost savings through such efficiencies.

Table 30: Estimated corporate revenues (annual profit margins) for the shellfish component of two MTA system options based on the operation of 7 farm sites. Revenues for an oyster and a scallop component are presented for adjacent and for integrated MTA.

Scallop Production Potential	**Shellfish Aquaculture System**	
	i-MTA	*a-MTA*
Net Site Revenue ($CDN):	$115,097.50	$22,255.75
Number of Sites	7	7
Total Annual Revenue:	**$805,682.50**	**$155,790.25**
Oyster Production Potential		
Net Site Revenue ($CDN):	$97,097.50	$4,255.75
Number of Sites	7	7
Total Annual Revenue:	**$679,682.50**	**$29,790.25**

Given the additional efficiencies offered through the management of multiple sites, it is highly probable that the net annual revenues presented in Table 30 represent very conservative estimates for such an operation. In fact, it is equally probable that net revenue values of 50-75% higher could be anticipated, bringing the estimated annual corporate profits at well over $1.0 million CDN.

Figure 83: Toxic phytoplankton (dinoflagellate) bloom off northwest Vancouver Island, August/2002.



The economic benefits of MTA are clearly optimized through the development and management of multiple farm sites. However, the risks associated with shellfish aquaculture (in general) are further reduced if multiple sites are proposed for such a corporate MTA development. In particular, the potential for harvest restrictions due to shellfish exposure to Harmful Algal Blooms (HAB's) can seriously disrupt market streams and potentially the unit pricing of these seafood products given such disruptions. Figure 83 shows the extent of such a bloom and how such seasonal impacts might result in specific coastal harvest restrictions. The image reveals a toxic dinoflagellate bloom that developed off the northwest side of Vancouver Island (late summer, 2002, and which subsequently was transported via tidal processes into the adjacent inlet system where shellfish and finfish aquaculture occur. The resulting harvest restrictions placed on scallop harvesting lasted 3 months.

The possibility of site closures (harvest restrictions) could also become a predictable occurrence at an MTA site given the potential impacts on tissue quality in response to use of antibiotics (or mother waterborne agents) within the multi-species system. The current research has demonstrated that such effects are probable, given the appropriate site conditions, but the resulting effects on tissue (and seafood) quality) are temporal in nature.

In managing multiple MTA sites the effects of area closures, whether a result of natural or process-related activities, could be effectively mitigated by compensating for harvest prohibitions by increasing harvesting in areas unaffected by the closures. This management approach remains transparent to seafood suppliers, distributors, retailers, and end-users (consumers), and would ensure an uninterrupted supply of shellfish product.

7.4 Social Benefits of a Finfish-Shellfish *Integrated*-MTA or *Adjacent*-MTA System

The economic and environmental benefits associated with Multi-Trophic Aquaculture (MTA) suggest that this approach may represent a valuable business development opportunity for finfish aquaculture companies looking to product diversification as a means for offsetting inherent fluctuations in single-product pricing and in product demand. However, in addition to the benefits that could be realized by individual companies, the introduction of MTA into temperate coastal regions may also have significant social benefits that result from the expansion of the shellfish aquaculture industry sector.

7.4.1 Present Industry Opportunities & Social Impact

Many of the marine sites currently producing salmon in coastal British Columbia are considered capable of supporting shellfish aquaculture. If we use the previous estimate of 45%, then a total of 56 sites could (hypothetically) support some form of finfish-shellfish MTA given the present status of the salmon aquaculture industry. Table 31 summarizes the economic and social impacts of such a development on the current shellfish aquaculture industry sector for this region.

Table 31: Estimated growth in shellfish industry sector given development of MTA at existing salmon aquaculture sites in coastal British Columbia. Direct, on-site jobs are estimated for *i*-MTA and *a*-MTA respectively. Estimate assumes that 45% of existing salmon farm sites are capable of supporting shellfish aquaculture.

Scallop Production Potential	**Shellfish Aquaculture System**
Number of Sites	56
Annual Production (MT):	5,559
Total On-Site Jobs:	224 to 336
Total Annual Sales:	$30,576,000.00
Oyster Production Potential	
Number of Sites	56
Annual Production (dozen):	11,827,200
Total On-Site Jobs:	448 to 560
Total Annual Sales:	$29,568,000.00

With the development of MTA at 45% (56 farms) of the existing salmon farm sites along the south coast of British Columbia, the direct benefits to the shellfish aquaculture sector, as a whole, are readily apparent. Increased employment is seen at the farm level with the creation of between 224 and 560 new jobs (estimates for *i*-MTA scallops to *a*-MTA oysters). Many (if not most) of these on-site jobs require a local contribution of labour (logistics), and thus provide the opportunity to rejuvenate coastal community employment levels that have historically fluctuated due to their association with the resource sector industries (fisheries, forestry). These new jobs represent full-time positions, are not seasonal in nature, and as such introduce a level of certainty and financial stability to such communities. In addition, the nature of the work offered at these aquaculture facilities requires a familiarity and experience with maritime life, an attribute share by the majority of residents in such remote areas of the coast.

The British Columbia Shellfish Growers Association (BCSGA) has estimated that this industry sector will grow from its present wholesale value of $20 million CDN to an anticipated $70 million by 2007 (www.bcsga.bc.ca). The industry association also predicts that this growth will be realized through more efficient use of existing tenure (intensifying production), with a short-term goal of achieving an average productivity of $20,000 CDN/tenure hectare (includes beach culture, which is currently the dominant method for production within the industry). The annual industry production of scallops contributed through MTA (Table??) could approach 5,560 MT with a farm-gate value of over $30.5 million CDN. For an oyster MTA industry, production could exceed 11.5 million dozen, with a value of over $29.5 million CDN.

The design of an MTA system, particularly for a truly *integrated* system, is necessarily based on an intensive culture design, representing a suspended aquaculture rather than a beach culture approach. Although the average salmon farm tenure is 20 hectares in size, the development of *i*-MTA would utilize a fraction of this area. For an *integrated*-MTA, the shellfish component is maintained within the finfish aquaculture infrastructure that, in total, occupies an area of approximately 2.4 hectares. Compared with the projected shellfish industry production levels of $20,000/hectare/annum, with a further goal of increasing efficiencies to produce $40,000/hectare/annum (BCSGA, 2003), the *i*-MTA system exemplified in this assessment would sustain shellfish production at over $225,000.00 CDN/hectare/year.

The large tenures used for system anchoring, and potentially for farm system rotation (fallowing), offers additional opportunities for shellfish development at a MTA site. In fact, the ability to develop both i-MTA and a-MTA system(s) at a finfish aquaculture site provides the opportunity of increasing this level of shellfish production by a factor of 2-4 times. Given this approach, economic and social benefits described herein would also be increased by this magnitude, and further suggesting that this evaluation provides a rather conservative estimation of economic and social benefits associated with such a regional development.

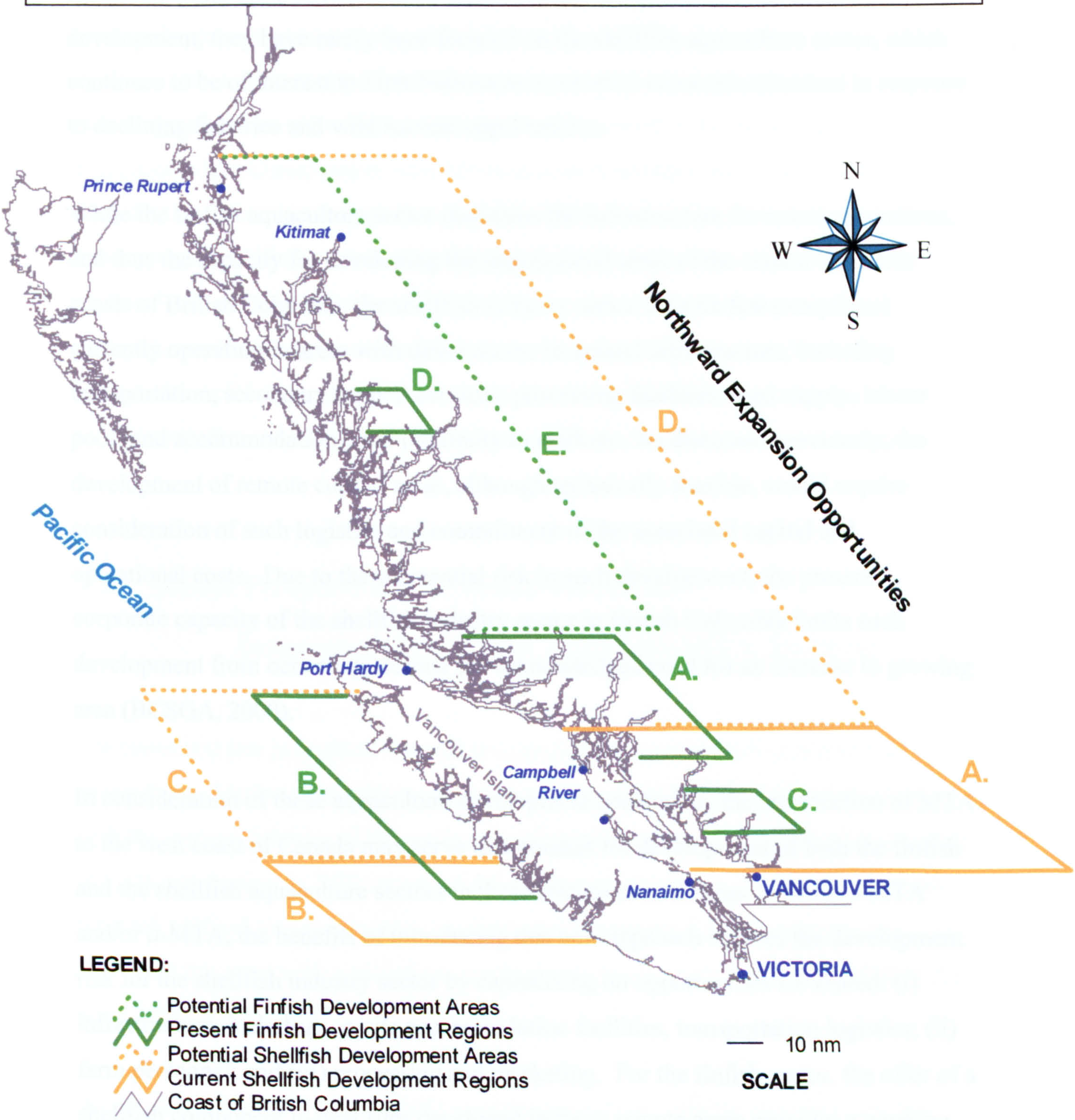
With a gross annual wholesale value of between $29 and 31 million CDN, the development of MTA at existing salmon farm sites could, with minimal improvements to current culture efficiency across the shellfish industry, achieve the industry production goal of $70 million by 2007. Furthermore, by introducing 56 new production sites, this productivity increase will result in substantially greater social benefits, through an increased employment base, than by the proposed focus on operational improvements to existing shellfish operations.

The social benefits of an expanded shellfish aquaculture sector, whether it occurs independently or as a result of policy change to permit MTA, are not limited to the direct employment opportunities offered at the site-specific level. Support services are typically diverse in the aquaculture industry, and are also sensitive to change in the productivity of the various aquaculture industry sectors. These support services include, for example, boat manufacturers and maintenance facilities, shellfish processing and ice plants, transportation companies, production equipment wholesalers (nets, rope, trays), engineering and environmental management consultants, marketing and promotional professionals, financial and educational institutions, etc. It is estimated that 4 indirect jobs are created for each additional direct job, which could result in an increase of between 896 and 2,240 indirect jobs. In total, MTA could add between 1,220 ands 2,800 industry-wide jobs. Many of these services are required in close proximity to the production tenures, and thus by increasing the productivity of the shellfish aquaculture sector, small and large coastal communities would benefit.

7.4.2 Industry Expansion Opportunities & Social Impact

Although the limitation of expansion space could be an important factor for exploring the feasibility of integrated multi-trophic aquaculture (i-MTA) in some temperate regions, the aquaculture development potential in western Canada is limited not by spatial constraint but rather by a combination of social and technical limitations. With aquaculture currently concentrated within the southern reaches of the coast (Figure ??), the central and northern regions remain virtually untouched.

Figure 84: Shellfish and finfish aquaculture expansion potential in coastal British Columbia. The current and potential aquaculture development regions, in relation to coastal population centers, are shown in green for finfish and orange for shellfish. A proportion of the sites within each of these areas will provide opportunities for MTA development.



Expansion of the finfish sector to the central and north coast is highly desirable by industry, but currently remains hindered by social uncertainty. Environmental issues (real or perceived) continue to serve as a deterrent for any new site, and coastal First Nations (native peoples) remain divided as to the risks of this sector as it relates to traditional coastal resources (e.g., wild salmon, shellfish beds, kelps, abalone). However, while these environmental issues remain in the forefront for finfish development, they have rarely been focused on the shellfish aquaculture sector, which continues to be of interest to First Nations as a potential economic stimulant in response to declining fisheries and wild harvest opportunities.

While the finfish aquaculture sector maintains the infrastructure for remote operations, and thus the capacity for developing the unpopulated areas of the central and north coasts of British Columbia, the shellfish industry sector is (with few exceptions) currently operating in areas with direct access to upland infrastructure, including transportation, secondary support services, processing facilities, seed supply, labour pool (and accommodation), and proximity to markets. As discussed previously, the development of remote coastal areas, although technically feasible, would require consideration of such logistics and commitment of the associated capital and operational costs. Due to the substantial risk in such development, the present corporate capacity of the shellfish industry sector in British Columbia limits such development from occurring, despite the acknowledged need for an increase in growing area (BCSGA, 2000).

In consideration of these aquaculture development constraints, the introduction of MTA to the west coast of Canada may serve as a conduit for development of both the finfish and the shellfish aquaculture sectors in these remote coastal areas. Whether *i*-MTA and/or *a*-MTA, the benefits of introducing this new approach reduces the development risk for the shellfish industry sector by capitalizing on opportunities for shared: (i) infrastructure, including on-site accommodation facilities, transportation logistics; (ii) farm personnel; and (iii) processing and marketing. For the finfish sector, the offer of a shellfish component to new sites developed in these remote areas provides a working opportunity for local communities, including First Nations, and an introduction to aquaculture that could build a working trust through demonstration of the realities associated with the environmental risks of these culture components.

Despite the inherent technical benefits of MTA, the addition of a shellfish (or other MTA) component may not necessarily be desirable to present-day finfish companies. For most of these multi-national producers the focus, albeit within a vertically-integrated corporate structure, remains on a single seafood product with the common argument of reduced complexity at all corporate levels, economy of scale, targeted knowledge base, etc. However, collaborative agreements to develop joint operations (*i*-MTA or *a*-MTA) may prove valuable from a social perspective. For example, while many British Columbia coastal First Nations continue to express concern over the development of finfish aquaculture within their traditional territories, citing resource conflicts and/or many of the perceived environmental risks misrepresented by eNGO's and by the media, they remain equally interested in pursuing opportunities offered through the shellfish aquaculture sector. In the wake of a declining wild fishery the need to revitalize coastal communities requires that such opportunities be realized.

7.5 Global Perspective on Finfish-Shellfish *Integrated*-MTA and *Adjacent*-MTA Systems

The continued rise in seafood demand has resulted in a corresponding increase in aquaculture. Marine aquaculture continues to increase its seafood production share with the capture fisheries of the world, revealing an average annual production increase of 5.1%; production in 2001 exceeded 5.0 million tonnes (FAO, 2002). The demand for fresh seafood also continues to increase, and the supply from aquaculture has addressed this need with a global per capita supply increase from 0.6 kg in 1970 to 2.3 kg in 2000 (FAO, 2002).

As concluded by FAO (2000, 2002) marine aquaculture, over the past 30 years, has (and continues) to expand, diversify and intensify globally. With technological and husbandry innovation this sector is providing ongoing potential for meeting increasing food demand, and as a result realizing economic benefits, increased trade, improvement in standard of living, and new opportunities for rural growth.

The introduction of Multi-Trophic Aquaculture (MTA) to temperate regions offers yet another avenue by which production levels can be increased to meet this global demand. In these environmentally conscience regions of the world, the application of MTA is an approach by which production increases can be achieved by reducing the environmental impact of the joint aquaculture activities. This, in itself, represents a very attractive social consideration, particularly in North America and in E.U. countries where environmental awareness and seafood safety issues are key development criteria.

The potential for integrating shellfish and finfish into productive MTA systems is similar for all temperate latitude countries that currently culture marine finfish species. As described for the salmon aquaculture industry of coastal British Columbia in this document, opportunities for integrating shellfish within the infrastructure and operational framework of existing finfish culture facilities in other temperate latitude countries should be similar. The well-established salmon aquaculture sectors in Norway, Chile, and Scotland could provide considerable potential for the co-culture of target shellfish species, resulting in a significant contribution (addition) to the shellfish production in each of these regions.

If we assume that finfish farms are, on average across all regions, producing 2,500 MT in a cage system configuration similar to that presented for the site analysed previously in this chapter, and that 45% of the farm sites in any of these global operational regions offer the biophysical attributes necessary to support finfish-shellfish MTA, as estimated for coastal British Columbia, then the potential economic and social value of this development, from a global perspective, can be estimated.

Table 32 presents an estimate of production capacity for an integrated shellfish component introduced to the salmon aquaculture sectors of temperate latitude countries, considering currently operating sites only. This assumes that all currently operating salmon farms comprise 2,500 MT steel cage facilities, defined as a '*typical farm site*', and that 45% of these sites are capable of supporting either oyster or scallop culture. It further assumes that the production cycle is fixed at 24 months, with the documented annual finfish production reflecting that generated from half of these farm sites. The use of global salmon industry status at the end of 2002 is employed in this estimation of i-MTA capacity.

Table 32: Global capacity for shellfish produced as an integrated MTA component at existing salmon farm facilities. Current capacity is projected using 2002 salmon production statistics from each region, and assumes that salmon is produced at a 'typical' site comprising 2,500 MT. The sites available to finfish-shellfish MTA are estimated at 45% of the total number of 'typical' sites within each of the major temperate regions.

Global *i*-MTA	**2002 Salmon Production *(MT x 1,000)***	**Estimated Number of 'Typical' Sites**	**Sites Available for MTA**	**Estimated Scallop Production (MT)**	**Estimated Oyster Production (dozen x 1000)**	**Projected Annual Shellfish Revenue (Farm-Gate) *Million USD***	**Projected Direct Jobs (average)**	**Projected Indirect Jobs (average)**
Norway	549	439	198	19,566	41,702	$90.21	1383	5534
Chile	354	283	127	12,617	26,890	$58.17	892	3568
Scotland	146	117	53	5,203	11,090	$23.99	368	1472
Canada-west	90	72	32	3,208	6,836	$14.79	227	907
Canada-east	42	34	15	1,497	3,190	$6.90	106	423
other (est.)	50	40	18	1,782	3,798	$8.22	126	504
Estimated Current Global *i*-MTA Capacity:				**43,873**	**93,507**	**$202.3**	**3,102**	**12,408**

Salmon Production Statistics:

Sloop, C. 2003. Chile fisheries products 2002 annual report.
USFDA Rep. C13022.
Scottish Executive. 2004. Farmed fish production survey for 2002.
(www.scottish.gov.uk).
Fisheries and Oceans Canada, Statistical Services. 2002 Canadian Aquaculture
Production Statistics. (www.dfo-mpo.gc.ca/statistics/aqua).
Statistics Norway. 2003. Fish Farming: Norway, Preliminary Figures.

The introduction of *i*-MTA systems at farm sites currently producing salmon would represent a significant contribution of marine seafood production to regional, as well as global aquaculture capacity. Table 32 suggests that scallop production of approximately 44,000 MT could be realized at these salmon farm sites, while application of an MTA approach using oysters could result in annual production of over 93.5 million dozen. The farm-gate value of such production would exceed $202 million USD, and provide in excess of 15,500 direct and indirect jobs.

Although this example provides a positive evaluation in terms of the current global potential for *i*-MTA development it is, at the same time, quite conservative in its estimate of the full, future potential for MTA. With an assumption of introducing shellfish to 45% of only the existing salmon farm sites, this does not account for the potential that could be realized through development of new MTA sites in each of these temperate regions. In addition, an increase in the global capacity for shellfish (as it relates to MTA development) could also be realized through the inclusion of *adjacent*-MTA systems to those configured for *integrated*-MTA. The impact of these expanded development scenarios are difficult to predict, but could quite easily result in economic and social benefits in the order of 5-10 times that estimated for the basic system development described above. Given this serious consideration of multi-trophic aquaculture systems in temperate waters, with a focus on finfish-shellfish and possibly macrophytes (Chopin, 2003), it is not improbable that shellfish production in such systems might contribute 0.25-0.50 million MT, globally, with a farm-gate value of between $1.0 and $2.0 billion USD. Increased employment would also become a significant benefit of such an expansion, resulting in 75,000 to 150,000 direct and indirect job opportunities.

In a global climate where the increased demand for food has accelerated the evolution of aquaculture, technological innovation plays an important role in meeting these production challenges while contributing to economic, social and environmental sustainability. Multi-Trophic Aquaculture has the potential to provide these sustainability attributes in coastal communities where the shift from a wild fishery to aquatic food production has become a reality.

The development of a balanced MTA could also add measurable environmental benefits to existing aquaculture systems, setting the stage for future production efficiencies and growth. Given a proper regulatory framework, including seafood (MTA products) and environmental quality surveillance, the potential water quality impacts on the shellfish component of a finfish-shellfish MTA (identified in this research initiative), and the associated risks over seafood safety, could be effectively managed to support this aquaculture evolution.

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