# ENRICHING EFFECTS OF SALMON FARMS IN BRITISH COLUMBIAN COASTAL WATERS AND THE INFLUENCE OF FLUSHING AND SEASONALITY.

By

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## ABSTRACT

Water samples at two salmon farms of diametrically opposed flushing characteristics in the Discovery Passage area were collected during the summer of 1988 at increasing distances downstream from the culture operations. The main objectives were to determine if salmon farms in this region are leading to elevated concentrations of total ammonia and dissolved organic carbon in the vicinities of the farms, resulting in higher concentrations of phytoplankton and bacteria.

Elevated surface total ammonia levels were observed in the immediate areas of the sites (*i.e.* < 10 m) at both locations, although the frequency of occurrence and the magnitude of the enrichment were greater at the area experiencing a weaker flushing regime. Chlorophyll a concentrations within the pens also appeared slightly higher compared to downstream levels during parts of the summer. The culture operations did not appear to have any effect on dissolved organic carbon and bacterial concentrations in the surrounding waters.

The ability to detect elevated levels of ammonium in the vicinity of fish farms was shown to be influenced by ambient levels of nutrients and phytoplankton biomass. The rapid reduction of ammonium concentrations within 25 m downstream of the culture facilities suggests that declines in water quality resulting from fish farming activity in this area seems unlikely. The findings of this study should be viewed as preliminary in nature given the limited size of the sampling program employed.

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# I. INTRODUCTION

Salmon farming in B.C. is growing at an almost explosive rate. In 1985 there were eight operating farms producing 107 tonnes of Chinook and Coho salmon. By 1988, the number of operating sites had increased over 15 fold to 125, while annual production grew to 6500 tonnes. Projections for 1990 indicate a further increase in the number of sites to over 150, with production doubling to 14000 tonnes per year (Fred Carpenter, Department of Fisheries and Oceans, pers. comm.). The Discovery Passage area located at the northern end of the Strait of Georgia is a region which exemplifies this tremendous growth of the salmon farming industry on the B.C. coast. Presently, 60 farms are in operation in this area, with 175 licenses for future sites currently awaiting approval (Rosenthal *et al.*, 1988).

Salmon farming unavoidably impacts to some degree on its immediate environment and the public generally views the increasing number of operating farms as a potential threat to marine water quality. Indeed, evidence for mariculture-induced water quality problems exist in nations such as Norway and Japan. Elevated levels of ammonium and decreases in dissolved oxygen concentrations in waters surrounding culture facilities have been noted in these countries along with the growth of their aquaculture industries (Arizono, 1978; Ervik *et al.*, 1985; Kadowaki and

Hirata, 1984.). The impact of salmon farms on marine waters is also of great importance to the farmers themselves, as both stress levels and disease incidences of cultured fish are enhanced by high levels of unionized ammonia and low dissolved oxygen concentrations.

The presence of salmon net-pens in marine waters can affect the surrounding physical, chemical and biological marine environment in a variety of ways. The potential environmental effects of salmon farms have been reviewed by Weston (1986), and include:

1) Changes in water circulation;
2) Sedimentation and accumulation of faeces and excess feed beneath the culture operation;
3) Changes in water chemistry;
4) Alteration of phytoplankton biomass and productivity;
5) Effects on abundance and species composition of benthic macrofauna;
6) Proliferation of bacteria pathogenic to humans and the effect of antibiotics on the surrounding biota;
7) Changes in species composition and abundance of fish and megafauna;
8) Disease transmission from cultured to wild stocks;
9) Introduction of exotic species and subsequent changes in the genetic fitness of wild stocks.

The overall goal of this study was to determine if the by-products associated with salmon farms in the Discovery Passage region are causing changes in the levels of primary and bacterial production in the immediate vicinity of the farms. The following introduction will therefore only review the portions of the literature which are pertinent to the influence of salmon farms on marine productivity. Information regarding the effects of fish culture facilities

on water circulation, nutrient levels, phytoplankton productivity and bacterial numbers will be presented. For a broader review of the environmental impacts of mariculture, the reader should refer to Rosenthal *et al.* (1988) and Weston (1986).

**A. Nutrient Loading**

There are 3 principal sources of nutrient loading related to salmon culture operations. The primary source is associated with the dispersion of the soluble end-products of salmonid protein metabolism which include total ammonia ($NH_3$ and $NH_4^+$), urea and phosphate. Ammonium forms the bulk of excretory nitrogen, although proportions of ammonium and urea can be variable (Gowen and Bradbury, 1987). Secondary nitrogenous end-products of salmonid metabolism such as nitrate and nitrite are produced through microbially mediated oxidation of ammonia and urea (Liao and Mayo, 1974). Although a small amount of phosphate is excreted in soluble form, the majority of phosphate waste is bound in the faeces and deposited on the sediments (Ennell and Lof, 1983).

Excretory products released from the fouling organisms bound to the rafts and net-pens of a culture operation constitute an important source of nutrient loading related to fish culture. The excretory products of mussels and other fouling organisms consist of total ammonia, amino-nitrogen, urea, and phosphate (Weston, 1986). The

substantial contribution of these invertebrates to nitrogen loading associated with fish farms was first documented by the presence of elevated total ammonia levels ($NH_3$ and $NH_4^+$) at inactive farm sites in Sechelt Inlet (Black and Carswell, 1986). The amount of loading from fouling organisms located on the supporting structures and nets of mariculture facilities is a function of the densities and growth rates of the organisms.

The third and least noteworthy source of nutrient loading from salmon farms results from the decomposition of excess feed and faeces deposited beneath the pens and the subsequent release of nutrients to the water column. Ammonium and phosphate are the principal breakdown products of feed and faeces and are found in high concentrations in the sediments and pore waters beneath mariculture facilities (Hall and Holby, 1986). The quantity of nutrients entering the water column from the sediment wastes is a complex function dependent on the area of waste dispersal, the reducing potential of the sediment and the flux rate at the sediment-water interface. The high settling velocities of uneaten food and faeces (0.06 to 0.15 $m \cdot s^{-1}$) preclude any substantial nutrient loss associated with microbial activity during settling (Gowen and Bradbury, 1987).

Dissolved inorganic nitrogen has long been held as the most important growth-limiting nutrient for phytoplankton in coastal marine waters (Dugdale, 1967) and increases in its concentration surrounding mariculture facilities could enhance phytoplankton and bacterial growth rates. Because phosphorus and silica in coastal environments are rarely found in growth-limiting concentrations, these nutrients can be considered inconsequential waste products of aquaculture facilities in most marine waters (Gowen and Bradbury, 1987). The remainder of this section on nutrient loading will therefore focus on the magnitude of nitrogen loading from marine salmon farms to the surrounding coastal waters.

There have been a few reported cases of increased ammonium levels in the vicinity of marine salmon farms. Ammonium concentrations surrounding net-pens in Norway were found to be 8-9 times higher than ambient levels (Ervik *et al.*, 1985). Consistently higher levels of ammonium in the vicinity of an experimental mariculture facility in Hendersen Inlet, Washington were noted during a period of limited mixing. Increased ammonium levels near the mariculture facility were observed at the surface but not at the near-bottom sampling depth indicating a negligible flux of regenerated ammonium from the sediment to the water column. Destratification of the water column after September and the resulting mixing prevented ammonium accumulation and depressions in dissolved oxygen levels near the salmon pens during fall, winter and spring

(Pease, 1977). Concentrations of total ammonia as high as 4 uM have been recorded at the surface within salmon pens at Sechelt Inlet, B.C., with maximal concentrations of 10 uM occurring at 6 m depth. Highest concentrations of total ammonia at 6 m were observed at mid-afternoon while surface maxima occurred during the early morning (S. Gormican, University of British Columbia, pers. comm.). This difference in the daily patterns of total ammonia concentrations at the surface and at 6 m suggests that ammonia levels are, in part, influenced by the nutritional requirements of the surrounding phytoplankton. Subsurface (3 m) levels of total ammonia within salmon pens located in a different area of Sechelt Inlet ranged from 0.2 to 0.9 uM higher than values recorded at a distance 27 m downcurrent from the enclosures (Black and Carswell, 1986). A lack of any increase in ammonium levels taken just above the bottom directly beneath the farms compared to ambient bottom water levels suggests very low rates of ammonium loading associated with decomposing waste feed and faeces in the sediment.

Estimates of the amount of nitrogen loading from salmon farms have been made by Gowen and Bradbury (1987). Their calculations are based on the assumption that 68 to 86% of the consumed nitrogen is voided from the fish as soluble ammonium and urea. Knowing the protein content of the feed, the digestibility of protein and the retention of nitrogen by the fish, the authors have estimated that 32 kg of

soluble ammonium are produced per tonne of food fed. A medium sized salmon farm (*ca.* biomass of 50 tonnes) feeding at a rate of 1% biomass/day would liberate 16 kg of soluble ammonium per day. Liao and Mayo (1974) have empirically derived ammonium production rates for pond-reared trout and estimated that these production rates follow the equation:

$$N_A = 0.0289 \cdot F,$$

where $N_A$ is the ammonium production rate at temperatures of 50-58 F, in pounds of $NH_4$-N per 100 lb fish per day and F is the feeding rate in pounds of food per 100 lb fish per day. This empirical estimation (28.9 kg of ammonium per tonne of food fed) is quite similar to that derived theoretically by Gowen and Bradbury (1987). Both estimates demonstrate that fish farms can provide substantial daily inputs of ammonium to the marine environment.

A considerable amount of nitrogen from the feed ends up in the sediment beneath the net-pens through the accumulation of waste feed and faeces. Approximately 20% of the feed used in marine salmon culture ends up on the bottom unutilized. Roughly 26% of the food eaten is excreted as faeces, 4% of which is comprised of nitrogen. Adding these numbers up, close to 30% of the nitrogen in the feed will end up on the bottom in particulate form (Gowen and Bradbury, 1987). A salmon farm with 50 tonnes of fish in the water uses roughly 500 kg of feed per day containing 40 kg of nitrogen. Thus, 12 kg of nitrogen per day are

deposited in the sediment below a large farm. Considering this particulate nitrogen is not diluted to anywhere near the same extent as its soluble counterparts, this represents a potentially significant source of nitrogen to the water column if the sediment-water exchange rate is high.

Elevated nitrogen levels in the sediments beneath salmon farms have been recorded at a number of different locations. The total nitrogen content of sediments beneath a mariculture facility in Henderson Inlet, Washington was approximately twice that of the reference area (Pease, 1977). Increased levels of ammonia nitrogen in interstitial and near-bottom water beneath salmon pens have also been noted in the Strait of Juan de Fuca (Weston, 1986) and in Sweden (Hall and Holby, 1986). The Swedish study revealed that ammonium concentrations in the first 5 cm of the pore water were over 1000 times greater than levels in pore water at the reference stations. *In situ* nitrification and denitrification of the total ammonia in the pore water and sediment could potentially act as a sink for the deposited particulate nitrogen. It has been clearly shown, however, that the oxidation of ammonia to nitrite and nitrate and the subsequent reduction of these compounds to nitrogen gas is virtually nonexistent in the anaerobic sediments commonly found below fish farms (Kaspar *et al.*, 1988). It is not surprising that levels of

nitrogen and ammonia in the sediments and pore waters beneath salmon farms have been found to be many times higher than background levels.

The quantity of nitrogen entering the water column from the enriched sediments below mariculture facilities can be determined through *in situ* direct measurements of solute fluxes across the sediment-water interface. Hall and Holby (1986) found ammonium fluxes in the sediments beneath salmon farms (0.0018 to 0.0198 g $NH_4$-N$\cdot m^{-2}\cdot day^{-1}$) to be 10-100 times higher than ammonium fluxes in ambient sediments. The enhancement of ammonium solute fluxes below fish farms results from the combination of increased levels of nitrogen in the sediments and lower rates of denitrification found in anaerobic sediments commonly seen beneath the culture operations (Ennell and Lof, 1983).

A theoretical estimation of ammonium loading from the sediments beneath salmon farms to the water column can be made knowing the ammonium solute flux and the area of enriched sediment. The area of sea bed over which waste feed and faeces will be dispersed is a function defined by the following relationship;

$$D = d\cdot V/v,$$

where D represents horizontal distance dispersed, d is water depth, V is current speed and v is settling velocity of the waste (Gowen and Bradbury, 1987). Thus a 50 tonne farm with

pens located 10 m above the bottom, using dry pellets settling at roughly 0.09 $m \cdot s^{-1}$, with prevailing currents of 0.1 $m \cdot s^{-1}$ and having an area of 3000 $m^2$ will disperse wastes over 5500 $m^2$. Using this affected area beneath a mariculture facility, the ammonium fluxes reported by Hall and Holby (1986) would deliver approximately 110 g $NH_4$-$N \cdot day^{-1}$ from the sediments to the water column. The amount of ammonium entering the water column from the sediments is almost 150 fold smaller than the 16 kg of soluble ammonium excreted daily by 50 tonnes of cultured fish.

The most significant input of nutrients from marine salmon farms results through the excretion of ammonium by the cultured fish. Although the size of the fish farm regulates the amount of ammonium entering the environment, the flushing capacity of the location will ultimately determine the rate of ammonium dilution and its final concentrations in the surrounding water.

**B. Changes in Phytoplankton Biomass, Production and Species Composition.**

Because inshore primary production is, for the most part, based on nitrogen in reduced forms such as ammonium and urea (Paasche, 1988), it is not unreasonable to assume that nitrogen loading from salmon farms could lead to increases in phytoplankton biomass and productivity. Furthermore, increased concentrations of nitrogen surrounding the farms could result in species composition

shifts through selection for algal groups with higher ammonium uptake rates and/or larger cell volumes better adapted for surviving in waters of higher trophic state (Harris, 1988).

There are a number of documented cases of freshwater fish farm effluents causing changes in ambient phytoplankton productivity. The effects of two large fish farms on primary production in a moderately sized (190 $km^2$) oligotrophic lake in Finland were investigated by Eloranta and Palomaki (1986). The authors discovered that areas 2 km downstream from the farms had chlorophyll a and primary production values two fold higher than stations located immediately upstream of the farms. A 50-100% increase in the number of phytoplankton species found in the more eutrophic waters in the area of the fish farms was also observed and attributed to the increased nutrient load from the farms. Higher levels of phytoplankton biomass and increased dominance of blue-green algal species were observed in the vicinity of a large trout culture operation in a small (48 ha) Polish eutrophic lake, and again, attributed to nutrient loading from the fish farm. Because the flushing capacity and circulation of these lakes is considerably less than that of most mariculture locations, it is difficult to interpret this information in relation to the enhancement of productivity around marine fish farms.

There is very little evidence to suggest that mariculture operations can cause changes in phytoplankton productivity in areas with even moderate flushing capacities. Increased concentrations of ammonium in waters surrounding salmon farms in Henderson (Pease, 1977) and Sechelt (Black and Carswell, 1986) Inlets were not accompanied by higher levels of chlorophyll a. A few documented cases of mariculture-induced enhancement of primary productivity do, however, exist. Arakawa (1973) correlated the higher frequency of phytoplankton blooms in Hiroshima Bay with increased levels of oyster production. Increases in the biomass and chlorophyll a content of the green alga, *Cladophora glomerata* were observed in areas immediately adjacent to fish farms located in the Baltic Sea (Ruokolahti, 1988). It should be noted, however, that because both the Sea of Japan and the Baltic Sea have relatively weaker flushing regimes and lower ambient nitrogen levels compared to the coastal areas and inlets of British Columbia, the effects of nutrient loading on primary production will be amplified in the former areas.

In laboratory experiments, Nishimura (1982) demonstrated that extracts from the faeces of cultured yellowtail tuna and mackerel meat used for feed caused significant increases in the growth rate of the red-tide forming dinoflagellate, *Gymnodinium nagasakiense*. In experiments designed to simulate conditions of seawater flowing over fish food and salmon faeces,

Parsons *et al.* (1989) found cell numbers of the heterotrophic dinoflagellate, *Oxyrrhis marina* to increase by a factor of 10 in the treated tanks compared to controls. No differences in diatom biomass or total primary production were noted between treated and control tanks. The experimental approaches of Nishimura (1982) and Parsons *et al.* (1989) both establish the possibility of phytoplankton species composition shifts in communities immediately above the deposited feed and faeces of a salmon farm. The effects of the soluble waste products of cultured fish on marine phytoplankton under controlled experimental condition has yet to be examined.

In summary, there is a limited amount of evidence to suggest that nutrient loading from the farms results in enhancement of primary productivity and changes in community structure. The magnitude of any potential changes will be a function of the nutritional state of the phytoplankton community, the ambient nutrient and light levels, natural patterns of algal succession, and most importantly, the flushing capacity of the location.

## C. Organic Carbon

Marine fish farms have a great potential to increase the levels of particulate and dissolved organic carbon in the surrounding water. The possible sources of carbon input from a culture operation to the water column include:

1) Bacterial decomposition of feed and faeces below the net pens resulting in the release of organic carbon from the sediment to the water column;
2) Leaching of carbon from feed and faeces as it sinks to the bottom;
3) Exudate release from phytoplankton and benthic algae associated with the fouling of nets and floats;
4) Bacterial decomposition of fouling organisms.

One of the most obvious effects of salmon farms on the marine environment is the accumulation of organic matter beneath the culture operation. Total carbon concentrations were found to be two-fold higher in sediments below salmon pens in Henderson Inlet (Pease, 1977), and a number of examples of total organic carbon enrichment have been reviewed by Weston (1986). While increases in the organic fraction of total carbon beneath salmon pens in Sechelt Inlet were noted, no changes in the percentage of volatile organic carbon in the same sediments were observed (Black and Carswell, 1986). The latter measurement can be used as an indication of the amount of organic material which remains to be leached from the sediments to the water column. Black and Carswell's (1986) investigation demonstrated that increases in the organic content of the sediment below fish farms do not necessarily lead to increases in the flux of carbon from the sediment to the

water column. In fact, although the presence of organically enriched sediments below culture operations is well documented and seems a likely source of organic carbon to the overlying water, there is no evidence to suggest an increased efflux of dissolved organic carbon from such deposits.

The leaching of carbon from sinking waste feed and faeces would seem a small if not negligible source of organic carbon to the surrounding water. The release of trace quantities of growth promoting substances such as vitamin $B_{12}$ from the feed could, however, have important effects on bacterial and algal growth. The negligible loss of carbon from feed through solution and microbial activity has been documented by Gowen and Bradbury (1987). Seawater tanks enriched with salmon feed and faeces, however, were shown to contain significantly higher levels of DOC (T.R. Parsons, pers. comm.).

The release of extracellular dissolved organic carbon from actively photosynthesizing phytoplankton is supported by a large body of literature (Storch and Saunders, 1978). In the event that increased rates of primary production exist in the immediate vicinity of culture operations, one would expect to see higher resulting levels of DOC. The large amounts of benthic algae residing on the nets and rafts of the fish farms would also be expected to provide an additional source of DOC to the surrounding water.

Decomposition of benthic algae and other fouling organisms could provide a final source of DOC and particulate carbon. Evidence to support the role of extracellular release of DOC and the decomposition of fouling organisms as important sources of organic carbon from fish farms is lacking.

**D. Bacteria and Antibiotics**

There is a general trend of increasing bacterial numbers and biomass with increasing levels of organic matter and primary productivity (Azam *et al.*, 1983). There have been many examples of elevated bacterioplankton concentrations in the vicinity of sources of organic carbon (Parsons *et al.*, 1988) and nutrients (Larsson and Hagstrom, 1982). Thus, the input of organic matter and nutrients associated with mariculture could reasonably be expected to lead to an increased number of bacteria produced in the vicinity of the culture sites (Rosenthal *et al.*, 1988).

The vast majority of studies regarding the influence of fish culture on bacterial biomass have measured the effects of culture activities on coliform and specifically, fecal coliform bacteria. The literature that documents increased levels of coliform and/or fecal coliform bacteria in the effluents of freshwater fish hatcheries and trout culture operations is accompanied by an equal number of cases where no effects of the culture operations are observed (Bergheim and Selmer-Olsen, 1978; Rosenthal *et al.*, 1988). In the marine environment, coliform levels in the vicinity of

salmon culture operations at Sechelt Inlet were found to be elevated compared to background concentrations. The fact that the enrichment was observed only at the near-surface sampling depth (3 m) suggests that the source of contamination was not associated with the unconsumed feed on the bottom (Black and Carswell, 1986).

In terms of bacterioplankton productivity, actual counts of bacterial numbers or total colony-forming units should give a more representative account of the bacterial consequences of organic enrichment associated with fish farms. Austin (1985) has conducted a number of surveys on freshwater fish culture operations and concluded that there was little effect of culture on bacterial numbers and composition. Investigation of the effluents of a coastal turbot-rearing facility, however, revealed an increase in bacterial numbers ranging from 3 to 50-fold. The large increases in bacterial biomass associated with the effluent of this marine culture operation were attributed to poor and unhygienic husbandry practices which resulted in increased levels of organic material and nutrients. Furthermore, direct comparisons concerning bacterial enrichment between net-pen culture and land-based rearing operations are suspect, as higher stocking densities and lower dilution capabilities associated with the latter form, will amplify the operations influence on bacterial production.

Experimental evidence suggests that solid fish farm waste can increase concentrations of bacterioplankton in the water directly above the sediment. Addition of salmonid feed and faeces to seawater tanks resulted in increases of bacterial numbers and heterotrophic uptake rates (Parsons *et al.*, 1989). Reduced levels of microflagellates in the treated tanks and the subsequent reduction in grazing pressure on the bacterioplankton were implicated as the causes of the elevated concentrations in the latter group.

Culture operations could also produce a poor environment for bacterial production in the surrounding waters through the employment of antibiotics and chemotherapeutics used in the treatment of fish disease. The amount of antibiotic released to the environment will be a function of the frequency and duration of treatment, the amount of antibiotic leaching from the feed, and finally, the persistence of the antibiotic in the sediment. Oxytetracycline (OTC) is the most commonly used antibiotic and is administered as a feed additive approximately 2-3 times over the summer months for the treatment of vibriosis and a host of other common fish diseases (Austin, 1985; Weston, 1986). The incidence of disease, and hence, the frequency of antibiotic usage, is dependent on prevailing water quality conditions and husbandry practices. Under poor conditions, the possibility of an almost continual administration of medicated feed during the warm months does exist.

The amount of OTC leaching from medicated feed has been found to be a positive function of water temperature, hydrogen ion concentration and surface/volume ratio of the pellets. Under conditions likely to be encountered in pond-rearing situations, up to 20% of the OTC in medicated feed is lost to the surrounding water within 15 minutes (Fribourgh *et al.*, 1969). Drug leaching in marine situations would be expected to be slightly less than 20% due to lower water temperatures and higher pH values. Marine bacteria responsible for sulphur and ammonium oxidation are unaffected by concentrations of OTC of up to 10 $mg \cdot l^{-1}$, however, a complete loss of activity occurs at concentrations of 100 $mg \cdot l^{-1}$ (Weston, 1986). Assuming an OTC leaching value of 20%, a culture operation with 50 tonnes of diseased fish, using 50-75 mg $OTC \cdot kg^{-1}$ body $weight \cdot day^{-1}$, would increase the OTC concentration in the immediate vicinity of the farm by 0.06 $mg \cdot l^{-1} \cdot day^{-1}$ if no water circulation existed around the farm. Given the large volumes of water that dilute marine fish farm effluents, it would seem very unlikely that the development of inhibitory OTC concentrations in the water column could occur through leaching alone.

The remainder of the OTC that does not leach from the waste feed will, of course, end up on the sediment below the net-pens. OTC has been found to be a relatively persistent drug in anoxic sediments commonly found below fish farms. The half-life of this agent has been determined through laboratory investigations to be approximately 10 weeks. The concentration of OTC measured in the sediments below a number of marine fish farms in Norway ranged from 0.1-4.9 $mg \cdot kg^{-1}$ dry matter. Antimicrobial effects within the sediments were estimated to occur for a period of up to 12 weeks after administration (Jacobsen and Berglind, 1988). The extent to which high levels of OTC and other antibiotics in the sediments contribute to water column concentrations has yet to be determined.

Direct evidence for the inhibition of bacterial production through the release of antibiotics from freshwater fish farms has been accumulated by Austin (1985). During periods of chemotherapy, bacterial numbers in the effluents were shown to be considerably less than within inflow waters. Dramatic increases in bacterial concentrations of the effluent waters were observed within a few days of the conclusion of chemotherapy. Reductions in bacterial numbers found in the effluents of fish farms undergoing disease treatments may also, in part, be a function of lower concentrations of organic matter related to reduced feeding regimes.

The increased levels of organic carbon and antibiotics associated with marine fish farms will have antagonistic effects on ambient bacterial levels. Coupled with the fact that bacteria compete with phytoplankton for growth-limiting nutrients such as ammonium (Harris, 1988), prediction of the influence of salmon farms on bacterial numbers becomes very complex. Direct measurements of bacterial concentrations in the vicinity of culture operations would therefore seem the most realistic and logical way of determining the consequences of fish culture on marine bacterial production.

**E. Water Circulation**

A recurrent theme throughout this discussion concerns the degree to which fish farm by-products are diluted by the surrounding water. The current regime at any site is critical in minimizing sedimentation and promoting the removal of by-products from the culture operation. A discussion of the extent to which the net-pens and raft structures of fish farms reduce current velocity in the culture structure and the surrounding area is therefore merited.

The alteration of current flow induced by a net-pen is dependent on such variables as mesh size, extent of fouling, stocking density, and the size and movement of fish within the pen. Studies on the effects of net-pens on current velocities by Inoue (1972) revealed that velocities within the pens were reduced to 35-81% of upstream values. Further

reductions in velocity occur for a series of nets aligned parallel to the current. After passage through 3 consecutive empty net-pens, current speeds ranging from 10-25% of the original upstream values can be expected. Because many B.C. salmon farms align up to 12 net-pens in a direction parallel to the prevailing currents, great reductions in flow at the downstream net-pens would be very likely.

Fluid dynamics principles have been used to estimate the effects of mariculture operations on water flow in the surrounding area (Weston, 1986). The distances to which a structure will effect the fluid flow surrounding it can be measured in diameters (dimension of a structure perpendicular to the direction of flow). For a porous structure such as a net-pen, current velocities should return to 95% of the original upstream velocities within one and two diameters from the structure at the upstream and sidestream areas, respectively. Downstream current velocities will be affected (*i.e.* < 95% of original current speed) for a distance of 20 diameters from the net-pens. Thus, two net-pens aligned perpendicular to the prevailing currents would affect downstream current velocities for at least 500 m. These values should be regarded as very rough approximations as the complex circulation patterns typical of coastal and estuarine environments confound attempts to estimate the influence of mariculture facilities on surrounding water circulation.

The fate and influence of salmon farm by-products such as ammonium and organic carbon on the marine environment is dependent on a number of biological and physical processes. The antagonistic nature of many of these processes warrants a circumspect attitude with respect to theoretical estimations of the effects of mariculture operations on marine waters. Direct measurements of the concentrations of salmon farm by-products and their effects on surrounding marine productivity in the immediate area of culture operations are justified.

# II. OBJECTIVES

The primary objective of this study was to determine if salmon farms in the Discovery Passage region are producing elevated concentrations of total ammonia and dissolved organic carbon in the immediate areas of the farms resulting in higher concentrations of bacteria and phytoplankton. By sampling at increasing distance from the farms, an estimate of the area of water showing any enriched characteristics could be realized.

To evaluate the influence of water circulation on the possible enriching effects of the culture operations, two sites of diametric flushing characteristics were sampled. One would expect to see stronger evidence for eutrophication or enrichment and a greater affected area at the site with poorer water exhange.

Finally, by sampling over a period of 3 months during the summer and early fall, the seasonal variability in any eutrophic effects produced by the farms could be estimated. Temperature, water column stratification and nutrient availability are important determinants for marine production and seasonal changes in these parameters will greatly influence any enriching effects of the culture operations. Changes in water quality and enhanced levels of production would most likely occur during periods when the photic zone is strongly stratified and depleted of nitrogen.

# III. MATERIALS AND METHODS

## A. Sampling Locations

The two farm sites chosen for this study are located on the islands of Quadra and Cortes. This area is bound to the north by Johnstone Strait and to the south by the Strait of Georgia (Fig. 1).

Yellow Island Aquaculture is found on the western side of Quadra Island adjacent to Seymour Narrows (Fig. 2A). This passage is renowned for its strong tidal flood currents which can reach speeds of up to 3 $m \cdot s^{-1}$ and result in intense mixing (Thompson, 1981). Mean tidal ranges from the Seymour Narrows reference station lie between 1.13 and 3.02 m (Anon., 1988). The salmon farm is located in a small bay of maximum depth of 48 m with the net-pen enclosures approximately 15 m above the bottom at low tide. Yellow Island Aquaculture produces 52 metric tonnes of Chinook, Coho and Steelhead annually.

Quartz Bay Sea Farms is located on the northern end of Cortes Island in a small, semi-enclosed bay bordered by Sutil Channel (Fig. 2B). Mean tidal ranges from the Sutil Channel reference station are between 1.22 and 3.44 m (Anon., 1988). Quartz Bay has a maximum depth of approximately 60 m with the culture facility located at the southern end of the bay 25 m above the bottom at low tide.

Figure 1. Locations of Yellow Island Aquaculture (Yellow Island) and Quartz Bay Sea Farms (Quartz Bay) in the Discovery Passage area.

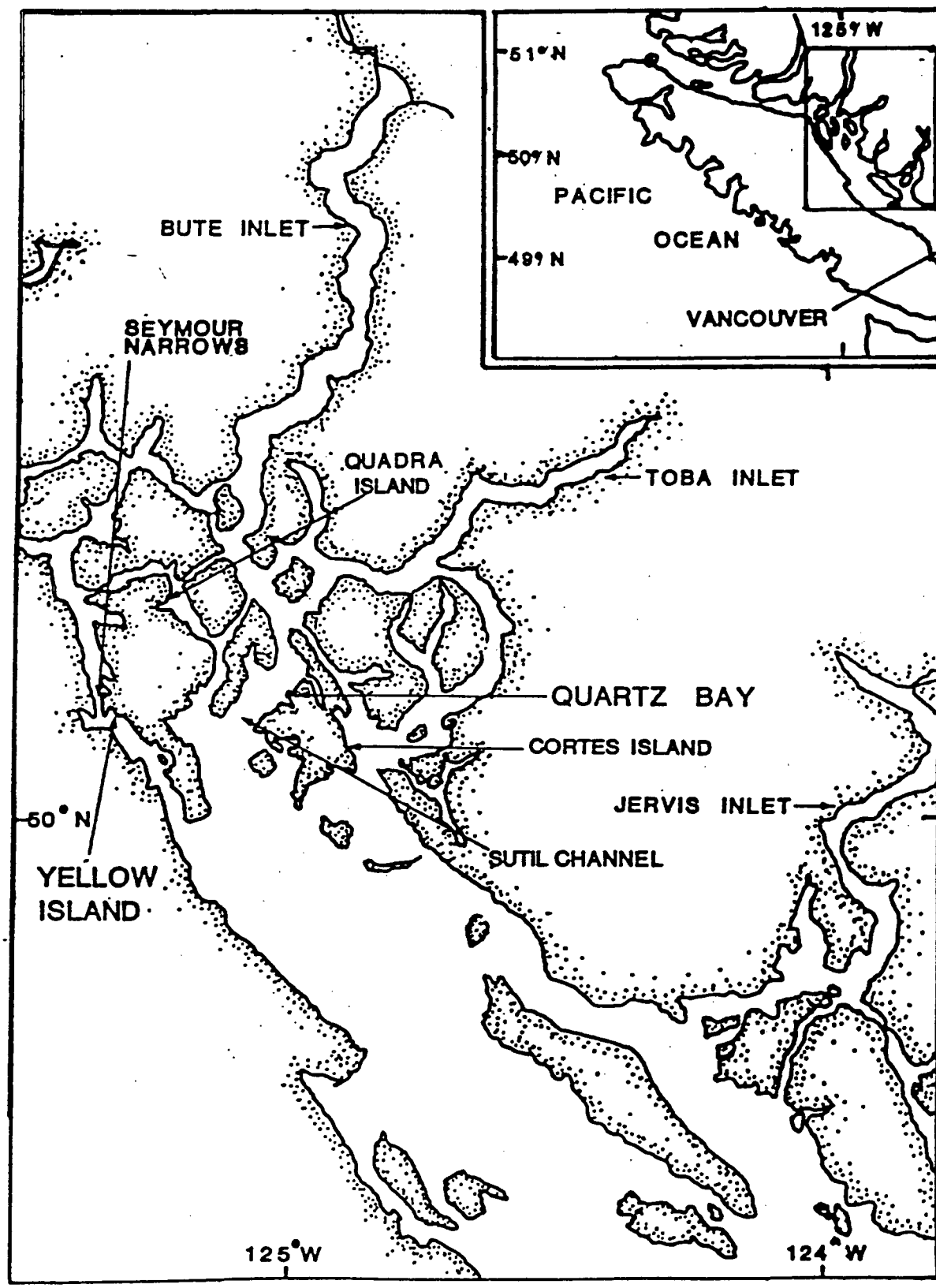
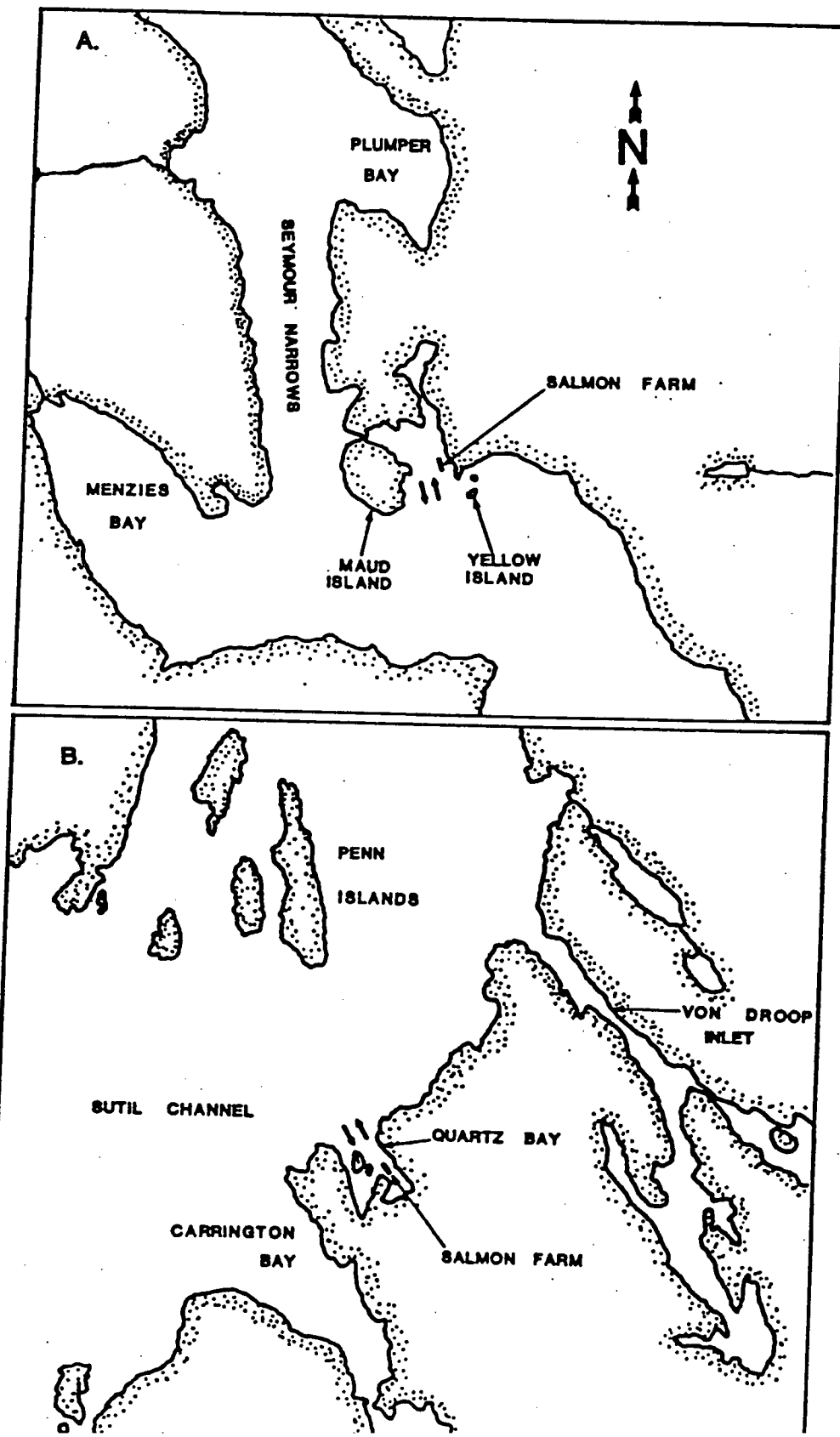


Figure 2. Detail of site locations for Yellow Island Aquaculture (A) and Quartz Bay Sea Farms (B) showing directions of ebb and flood currents.



Quartz Bay Sea Farms has an annual production of *ca.* 65 tonnes of Chinook and Coho.

Seawater samples from both farms were taken just below the surface (0.5 m) with a plastic bucket within the net-pens and at 3, 10 and 25 m from the pens corresponding to stations 0, 1, 2 and 3, respectively. In order to maintain a constant distance between the stations and the culture facility during all sampling periods, a polypropylene line was fixed from the rafts to the surrounding boom structure and marked at the appropriate distances. Samples were taken from the downcurrent side of the enclosures along the transect line which was strung parallel to the ebb current direction. Concentrations of total ammonia ($NH_3$ and $NH_4^+$), nitrate and nitrite ($NO_3^-$ and $NO_2^-$), chlorophyll a and bacteria were determined as well as primary productivity, phytoplankton species composition and dissolved organic carbon (DOC) absorbance at each station.

The Yellow Island Aquaculture site was sampled four days over the summer between late June and September. In order to evaluate the influence of tidal displacement on the dilution and concentrations of the various water quality parameters examined, the Yellow Island site was sampled two times per day as close to low and high tides as possible (Table 1).

The difficulty and expense of accessing Quartz Bay Sea Farms restricted the sampling frequency to three periods over the summer between early July and September. In order to minimize the time from sample collection to the initial processing and freezing which occurred at Yellow Island, only one daily set of samples was taken at Quartz Bay. The time of sampling ranged from 10:00 to 12:00 corresponding to low and midwater tidal heights (Table 1).

Table 1. Sampling frequency and corresponding tidal position.

| SAMPLING LOCATION | SAMPLING DATE | SAMPLING TIME | TIDAL POSITION |
|---|---|---|---|
| Yellow Island | 30/06/88 | 11:00 | Low Tide |
| | | 16:00 | High Tide |
| | 06/08/88 | 10:00 | Midwater |
| | | 14:30 | High Tide |
| | 07/08/88 | 10:00 | Midwater |
| | | 14:30 | High Tide |
| | 21/09/88 | 9:00 | Low Tide |
| | | 13:00 | High Tide |
| Quartz Bay | 01/07/88 | 12:00 | Low Tide |
| | 08/08/88 | 11:00 | Midwater |
| | 20/09/88 | 10:00 | Midwater |

**B. Physical Measurements**

Temperature/salinity (T/S) profiles were taken at both sites (Station 1) following the first sample collection with an AUTOLAB$^{TM}$ model 602 portable temperature/salinity probe. The water column was sampled at interval depths of 2 m. The

instrument was calibrated prior to the field season according to the methods described in the AUTOLAB$^{TM}$ user handbook. The temperature and salinity precisions of the instrument correspond to $\pm$ 0.1 C and $\pm$ 0.03 $^{o}/_{oo}$. In order to correct for drift of the T/S probe over the summer, the temperature of a surface sample was determined with a glass thermometer and a salinity sample was collected for analysis with a laboratory salinometer following each profile. Differences between the surface T/S values determined with the probe and those measured manually or in the laboratory were used to calculate correction factors. The original T/S values were then adjusted through multiplication with the corresponding correction factors. Sigma-t values were calculated from the raw data according to the International Equation of State of Seawater, 1980 at one atmosphere pressure (Pond and Pickard, 1982).

Surface current speeds and directions were determined using a 1 X 1.5 m window-blind drogue of similar construction to those described by Buckley and Pond (1976). The drogue was deployed from the rafts surrounding the pens and was attached to the marked transect line with a short leash and brass clip. Current speed was calculated from the mean of two 25 m trials. Surface current speeds and directions were measured immediately prior to the collection of seawater samples.

Water clarity was estimated using a 30 cm white metal Secchi disc lowered in the shade of the rafts as close to mid-day as possible. Extinction coefficients were calculated based on the equation,

$$K' = 1.7/D_s,$$

where K' is the extinction coefficient of the water and $D_s$ is the Secchi disc depth in meters (Parsons *et al.*, 1977).

**C. Total Ammonia and Nitrate**

Seawater samples for nutrient determinations were collected from the sampling bucket in 500 ml Nalgene bottles. The samples were filtered under a 0.5 atmospheric pressure vacuum through 47 mm AA Millipore membrane filters. The filtrate was collected in 50 ml glass test tubes suspended in the filtration flasks with string and a 30 ml subsample was then frozen (-20 C). In order to estimate the amount of variability induced in the total ammonia samples due to the effects of freezing, 30 ml of a 1.5 *u*M $NH_4Cl$ solution prepared in 3% NaCl was frozen along with every set of nutrient samples. The maximum length of time from collection to freezing over the course of the summer ranged from one to three hours for Yellow Island and Quartz Bay, respectively. All Nalgene bottles and glassware used in the collection and filtration of nutrient samples were acid washed in 10% HCl and rinsed three times in distilled/deionized water.

Nitrate ($NO_3^-$ and $NO_2^-$) and total ammonia ($NH_3$ and $NH_4^+$) concentrations were determined following the procedures of Wood *et al.* (1967) and Slawyk and MacIsaac (1972), respectively. All nutrients were measured using a Technicon Autoanalyzer$^{TM}$ II. The precision of the nitrate analysis at the 20 *u*M level has been determined by Parsons *et al.* (1984) as $\pm$ 0.5 *u*M. The precision of the total ammonia technique was determined experimentally by analyzing the same 20 samples in two different runs. Precision at the 2 *u*M level was calculated as $\pm$ 0.02 *u*M.

**D. Chlorophyll a**

All seawater samples for chlorophyll a determinations were collected in 1 l Nalgene bottles. Seawater volumes ranging from 0.75 to 1 l were filtered through 47 mm AA Millipore membrane filters under a 0.5 atmospheric pressure vacuum with the addition of 3-5 drops of $MgCO_3$ solution. The samples retained on the filter paper were wrapped in wax and aluminum paper, placed in a dark plastic container with Drierite$^{TM}$ and frozen at -20 C.

The experimental procedures and spectrophotometric equations used for the chlorophyll a analyses are described by Parsons *et al.* (1984). A Spectronic 21$^{TM}$ spectrophotometer with 50 mm path-length cuvettes was used to determine the absorptions at the specified wavelengths. Precision at the 5 mg/m$^3$ level has been determined as $\pm$ 0.21 mg/m$^3$ chlorophyll a.

**E. Primary Production**

Primary productivity was calculated by measuring the uptake of radioactive carbon with *in situ* incubations. Precision at the 30 mg $C \cdot m^{-3} \cdot hr^{-1}$ level for a 3 hr incubation using 5 *u*Ci has been determined as $\pm$ 3 mg $C \cdot m^{-3} \cdot hr^{-1}$ (Parsons *et al.*, 1984). Seawater samples were collected from the sampling bucket in 500 ml Nalgene bottles and divided into one opaque and two clear 125 ml BOD bottles. The samples were then innoculated with 5 *u*Ci of radioactive carbonate. The bottles were placed in a large mesh goodie bag containing 12 small compartments and submerged to a depth of 0.5 m at Station 1. The incubations ranged from 2 to 2.5 hrs in length and occurred during mid-day following the morning sampling period. While Yellow Island seawater samples were innoculated almost immediately after collection, the samples from Quartz Bay had to be transported back to Yellow Island prior to innoculation and incubation. Quartz Bay samples were innoculated approximately 2 hours after collection and incubated at Yellow Island, Station 1.

Following incubation the samples were filtered at a 1/3 atmospheric pressure vacuum through 47 mm AA Millipore membrane filters. The filters were placed in vials containing 10 ml of Aquasol scintillation cocktail and stored in the dark until the radioactivity of the samples could be determined using an Isocap/300$^{TM}$ liquid scintillation counter. The standard error of the primary

production values was calculated based on the standard deviation of the mean of the two clear bottle replicates.

Primary production rates were divided by the corresponding chlorophyll a values to produce production/biomass (P/B) ratios. This primary productivity index has been established as a more accurate measure of comparing various photosynthetic rates (Lorenzen, 1963).

**F. Phytoplankton Species Composition**

For determining phytoplankton species composition, 150 ml samples were taken from the sampling bucket and preserved in 250 ml glass bottles with 10-15 drops of Lugol's solution. The samples were settled in 10 ml plankton chambers for at least 4 hrs prior to counting. The phytoplankton was scanned at a magnification of 94.5 X (low power) on a Zeiss inverted microscope. The entire chamber bottom was viewed and the percent species dominance was estimated based on visual observation rather than cell counts. In addition, the ten most abundant species were ranked, from visual estimates of relative biomass.

## G. Dissolved Organic Carbon

Relative dissolved organic carbon concentrations were estimated from the absorption of filtered seawater samples at 280 nm. Both Parsons *et al.* (1984) and Krom and Sholkovitz (1977) have shown that DOC absorbance at 280 nm is proportional to estimates of its concentration obtained by dry combustion methods. Although no concentrations of DOC were determined in this study, absorbance was used to get a relative measure of DOC concentrations.

During the filtration of the chlorophyll a seawater samples, 50 ml of filtrate was collected in a test tube suspended in the filtration flask. The filtrate was frozen (-20 C) in 100 ml glass bottles. In the laboratory, the samples were thawed and absorption at 280 nm was measured using a Coleman$^{TM}$ 124D double beam spectrophotometer with 100 mm path-length cuvettes.

## H. Bacteria

Direct counting by fluorescence microscopy was used to determine bacterial numbers with a precision at the $10 \times 10^5$ cells$\cdot$ml$^{-1}$ level of $\pm$ $2 \times 10^5$ cells$\cdot$ml$^{-1}$ (Parsons *et al.*, 1984). To collect bacterial samples, 10 ml of seawater was withdrawn from the sampling bucket and injected into a scintillation vial containing 1 ml of filtered 40% formaldehyde. The samples were refrigerated at 5 C in the dark prior to counting. To determine bacterial concentrations, one ml subsamples were dyed with acridine

orange and filtered through a previously stained Nuclepore filter. Bacterial numbers were estimated from the mean of 10 fields counted at 400 X using a Zeiss standard 18 microscope. Standard errors were calculated based on the standard deviation of the mean of 10 fields.

## I. Data Analysis

All calculations used for the determinations of sigma-t values, chlorophyll a concentrations and primary productivity rates were performed using the LOTUS$^{TM}$ 1-2-3 spreadsheet program. The SYSTAT$^{TM}$ data package was used for all the statistical tests and correlations needed in this study.

In order to test for statistical differences in the various water quality and production parameters between stations at each location, data from the entire summer for each parameter needed to be combined to increase sample size. Sample sizes used in the nonparametric test for determining differences between station for various parameters are summarized in Table 2.

Table 2. A summary of the sampling frequencies and sizes used in the statistical tests to determine differences between stations for the various water quality parameters studied. ( )* refers to primary productivity values determined only once per day.

| LOCATION | SAMPLINGS/DAY | SAMPLING DAYS OVER SUMMER | SAMPLE SIZE (n) |
|---|---|---|---|
| Yellow Island | 2 (1)* | 4 | n=32 (8)* |
| Quartz Bay | 1 | 3 | n=12 |

The small sample sizes ($8 \leq n \leq 32$) made it impossible to determine if the statistical distributions for each parameter were normal and homoscedastic. In light of this, the nonparametric Kruskal-Wallis single factor analysis of variance test (Sokal and Rohlf, 1981) was used to determine if there were any significant differences between stations at each location for concentrations of total ammonia, chlorophyll a, DOC and bacteria as well as primary production rates and productivity indices.

In order to determine if the levels of the examined parameters were significantly different in the immediate vicinity of the pens compared to ambient levels, the data were divided into inner (mean values of Stations 0 and 1) and outer (mean values of Stations 2 and 3) groups. This analysis was performed because similarities in the concentrations of various parameters within the outer and inner station groupings would statistically mask any differences between all stations if they were compared

individually. The Mann-Whitney nonparametric two-sample rank sum test (Sokl and Rolf, 1981) was employed to test for differences between these two groupings.

Two matrices of Spearman correlation coefficients were constructed to determine the relationships among the physical and biological parameters at Yellow Island and Quartz Bay. To do this, data collected over the entire summer from all stations were lumped together at each location. The resulting correlations therefore reflect the effect of seasonally influenced physical variables such as surface salinity and water column stratification on the biological variables of interest.

# IV. RESULTS

## A. Physical Studies

A comparison of the sigma-t profiles of Yellow Island and Quartz Bay reveals a marked difference in the water column stability at the two locations (Fig.'s 3A and 3B). Yellow Island appears to be a well mixed site as no pycnocline developed over the course of the summer. The Quartz Bay profile clearly demonstrates the presence of a strong pycnocline at 4 m depth during the middle and late summer. In order to quantify the magnitudes of stratification at the two locations, the differences in sigma-t values at the surface and 10 m depth were calculated to produce a stratification parameter as follows,

$$\text{Stratification} = (\text{sigma-t}_{10} - \text{sigma-t}_{0})/\ 10\ \text{m},$$

where sigma-$t_{10}$ and sigma-$t_0$ are the respective 10 m depth and surface values. The higher the stratification value, the greater the degree of stability in the water column. The stratification values for Yellow Island and Quartz Bay are presented in Tables 3 and 4, respectively.

Figure 3. Sigma-t profiles at Yellow Island (A) and Quartz Bay (B), summer, 1988.

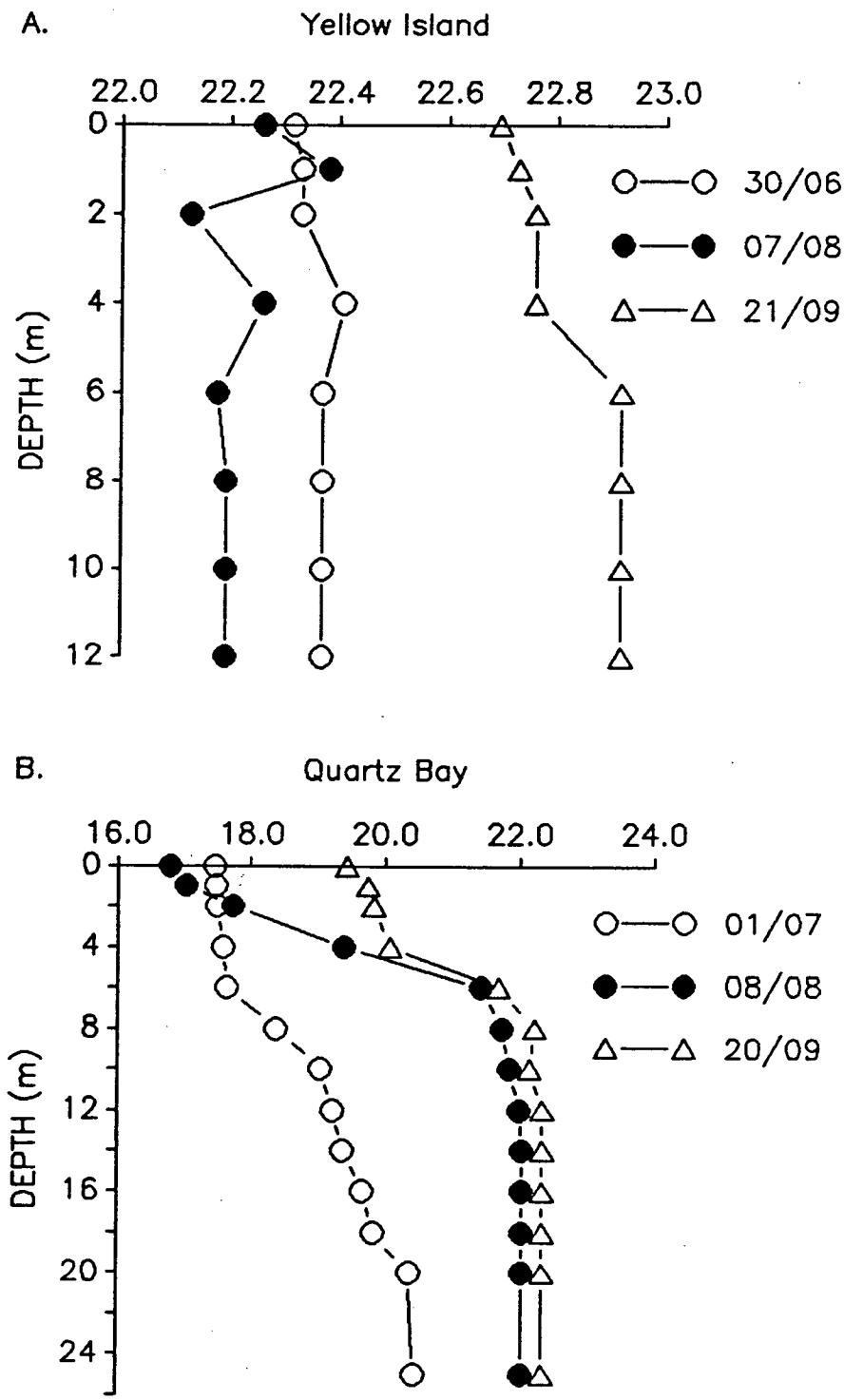


Table 3. Summary of physical data for Yellow Island between June and September, 1988. Stratification was determined as the average change in sigma-t in the top 10 m.

| PARAMETER | TIDAL POSITION | SAMPLING DATES 30/06 | 06/08 | 07/08 | 21/09 |
|---|---|---|---|---|---|
| Sfc. Temp. ( C ) | Mid | 10.6 | 10.8 | 11.6 | 10.7 |
| Sfc. Salinity ( $^{o}/_{oo}$ ) | Mid | 29.18 | 29.24 | 29.33 | 29.69 |
| Stratification (sigma-t units$\cdot m^{-1}$) | Mid | .005 | -.005 | -.107 | .022 |
| Secchi Depth ( m ) | Mid | 11.0 | 7.5 | 7.0 | 10.0 |
| Extinction Coefficient ($m^{-1}$) | | 0.15 | 0.23 | 0.24 | 0.17 |
| Compensation Depth ( m ) | | 30 | 20 | 19 | 27 |
| Sfc. Current Speed ( $cm \cdot s^{-1}$) | Low | 17.5 | 4.8 | 5.6 | 7.1 |
| | High | 6.5 | 37.5 | 20.6 | 18.2 |

Table 4. Summary of physical data for Quartz Bay between July and September, 1988. Stratification was determined as the average change in sigma-t in the top 10 m.

| PARAMETER | TIDAL POSITION | SAMPLING DATES 01/07 | 08/08 | 20/09 |
|---|---|---|---|---|
| Sfc. Temp. ( C ) | Mid | 15.8 | 14.3 | 10.9 |
| Sfc. Salinity ( $^{o}/_{oo}$ ) | Mid | 24.13 | 22.87 | 25.54 |
| Stratification (sigma-t units$\cdot m^{-1}$) | Mid | .161 | .583 | .272 |
| Secchi Depth ( m ) | Mid | 9.0 | 5.0 | 5.5 |
| Extinction Coefficient ($m^{-1}$) | | 0.19 | 0.34 | 0.31 |
| Compensation Depth ( m ) | | 24 | 13 | 15 |
| Sfc. Current Speed ( $cm \cdot s^{-1}$ ) | Mid | 7.5 | 4.0 | 11.0 |

Yellow Island had summertime surface temperature and salinity averages of 10.9 C and 29.36 $^{o}/_{oo}$ compared to 13.7 C and 24.18 $^{o}/_{oo}$ summertime averages at Quartz Bay. The Quartz Bay location experienced higher variability in both surface temperature and salinity values over the course of the summer ( Tables 3 and 4 ).

Speeds from the current drogue studies ranged from 6.5 to 37.5 $cm \cdot sec^{-1}$ at Yellow Island and 4.0 to 11.0 $cm \cdot sec^{-1}$ at Quartz Bay. The drogues generally travelled parallel to the expected ebb tide directions which were southwesterly at Yellow Island and northwesterly at Quartz Bay.

The Secchi disc depths and corresponding extinction coefficients for Yellow Island and Quartz Bay are summarized in Tables 3 and 4, respectively. It appears that at both locations, extinction coefficients reach a maximum in August with the Yellow Island summertime average ($k' = .20\ m^{-1}$) being considerably lower than that of Quartz Bay ($k' = .28\ m^{-1}$). In order to get an estimate of the compensation depth at each location, the extinction coefficients were used in the following equation,

$$I_d = I_o \cdot e^{-k.d} ,$$

where $I_d$ and $I_o$ are the respective light intensities at depth d and the surface, and k is the extinction coefficient of the water (Parsons *et al.*, 1977). At the compensation depth ($D_c$), the ratio of $I_d/I_o$ can be approximated as .01

and the preceding equation reduces to,

$$D_c = -\ln(.01/k).$$

Calculated compensation depths for Yellow Island and Quartz Bay are given in Tables 3 and 4.

**B. Total Ammonia and Nitrate**

The surface concentration of total ammonia at Yellow Island within the net-pens (Stn. 0) ranged from 1.15 to 3.03 *u*M between late June and August. Surface concentrations at the outer station (Stn. 3) over the same time period had a larger range of 1.02 to 4.50 *u*M (Table 5). The spatial distributions of total ammonia concentrations at Yellow Island during the four sampling periods are shown in Figures 4A, 5A and 6A. The only evidence of elevated total ammonia concentrations in the immediate vicinity of the net-pens was seen during the low tide sampling period on June 30$^{th}$ (Fig. 4A).

Total ammonia at Quartz Bay within the net-pens and at the outer station ranged from 2.00 to 5.59 *u*M and 1.14 to 3.40 *u*M, respectively (Table 6). Total ammonia values for the inner two station at Quartz Bay are noticeably higher than the equivalent Yellow Island values. A clear pattern of elevated total ammonia concentrations in the immediate vicinity of the net-pens is seen in early July and late September (Fig. 7A).

Table 5. Summary of water quality data for Yellow Island, summer, 1988. (-) denotes missing values.

| PARAMETER | STATION | SAMPLING DATES 30/06 | 06/08 | 07/08 | 21/09 |
|---|---|---|---|---|---|
| TOTAL AMMONIA | 0 | 3.03 | 2.40 | - | 2.54 |
| ( uM ) | 1 | 2.48 | 2.45 | - | 2.08 |
| | 2 | 1.79 | 1.98 | - | 3.71 |
| Low Tide | 3 | 1.02 | 2.89 | - | 4.50 |
| | 0 | 2.40 | 1.15 | 1.80 | 1.75 |
| High Tide | 1 | 2.08 | 2.98 | 2.35 | 1.94 |
| | 2 | 2.05 | 2.49 | 2.64 | 3.16 |
| | 3 | 1.27 | 1.60 | 2.22 | 2.27 |
| CHLOROPHYLL A | 0 | 4.36 | 4.53 | 4.53 | 3.02 |
| ( mg Chl $a \cdot m^{-3}$ ) | 1 | 1.34 | 3.75 | 4.08 | 2.89 |
| | 2 | 2.02 | 4.73 | 4.17 | 3.98 |
| Low Tide | 3 | 1.74 | 5.06 | 4.19 | 3.39 |
| | 0 | 3.28 | 4.73 | 2.35 | 2.92 |
| High Tide | 1 | 1.98 | 3.67 | 3.06 | 3.01 |
| | 2 | 1.66 | 3.98 | 3.93 | 3.28 |
| | 3 | 1.63 | 3.86 | 3.86 | 2.73 |
| P/B | 0 | 0.80 | 1.84 | 0.59 | 1.33 |
| ($mg_1C \cdot mg \cdot Chl\ a^{-1}$. | 1 | 2.19 | 0.80 | 1.02 | 2.87 |
| $hr^{-1}$) | 2 | 1.14 | 1.61 | 0.80 | 0.54 |
| Low Tide | 3 | 1.27 | 0.92 | 1.00 | 0.26 |
| BACTERIA | 0 | 5.42 | 6.04 | 7.50 | 3.70 |
| ($cells \cdot ml^{-1} x 10^5$) | 1 | 6.34 | 3.48 | 8.44 | 2.46 |
| | 2 | 5.96 | 6.78 | 9.92 | 3.36 |
| Low Tide | 3 | 6.64 | 5.10 | 7.26 | 4.12 |
| | 0 | 5.04 | 4.44 | 5.42 | 3.86 |
| High Tide | 1 | 5.38 | 6.34 | 5.52 | 3.56 |
| | 2 | 5.58 | 6.98 | 6.32 | 3.60 |
| | 3 | 4.90 | 6.40 | 7.52 | 4.02 |
| DOC ABSORBANCE | 0 | .090 | .146 | .132 | .153 |
| ( 280 nm ) | 1 | .116 | .158 | .132 | .188 |
| | 2 | .253 | .253 | .161 | .216 |
| Low Tide | 3 | .187 | .253 | .234 | .106 |
| | 0 | .211 | .172 | .111 | .106 |
| High Tide | 1 | .151 | .166 | .204 | .077 |
| | 2 | .209 | .129 | .167 | .138 |
| | 3 | .195 | .192 | .168 | .140 |

Table 6. Summary of water quality data for Quartz Bay, summer '88. All samples were taken between low and mid tide.

| PARAMETER | STATION | SAMPLING DATES 01/07 | 08/08 | 20/09 |
|---|---|---|---|---|
| TOTAL AMMONIA ( uM ) | 0 | 5.59 | 2.00 | 5.17 |
| | 1 | 5.34 | 2.63 | 4.57 |
| | 2 | 2.74 | 1.63 | 1.90 |
| | 3 | 3.40 | 1.14 | 2.15 |
| CHLOROPHYLL A ( mg Chl $a \cdot m^{-3}$ ) | 0 | 5.06 | 8.47 | 3.58 |
| | 1 | 3.76 | 7.21 | 3.37 |
| | 2 | 2.39 | 6.51 | 4.89 |
| | 3 | 3.00 | 6.77 | 5.21 |
| P/B ($mg_1 C \cdot mg$ Chl $a^{-1} \cdot hr^{-1}$) | 0 | 1.57 | 1.67 | 2.10 |
| | 1 | 1.05 | 1.66 | 1.62 |
| | 2 | 1.31 | 1.26 | 2.33 |
| | 3 | 1.46 | 1.57 | 1.97 |
| BACTERIA ($cells \cdot ml^{-1} x 10^5$) | 0 | 9.20 | 13.22 | 7.42 |
| | 1 | 14.84 | 17.38 | 5.20 |
| | 2 | 18.40 | 14.98 | 6.50 |
| | 3 | 17.60 | 15.28 | 5.24 |
| DOC ABSORBANCE ( 280 nm ) | 0 | .224 | .150 | .178 |
| | 1 | .252 | .166 | .161 |
| | 2 | .239 | .105 | .111 |
| | 3 | .206 | .134 | .131 |

Figure 4. Total ammonia (A) and chlorophyll a (B) concentrations at Yellow Island, late June, 1988.

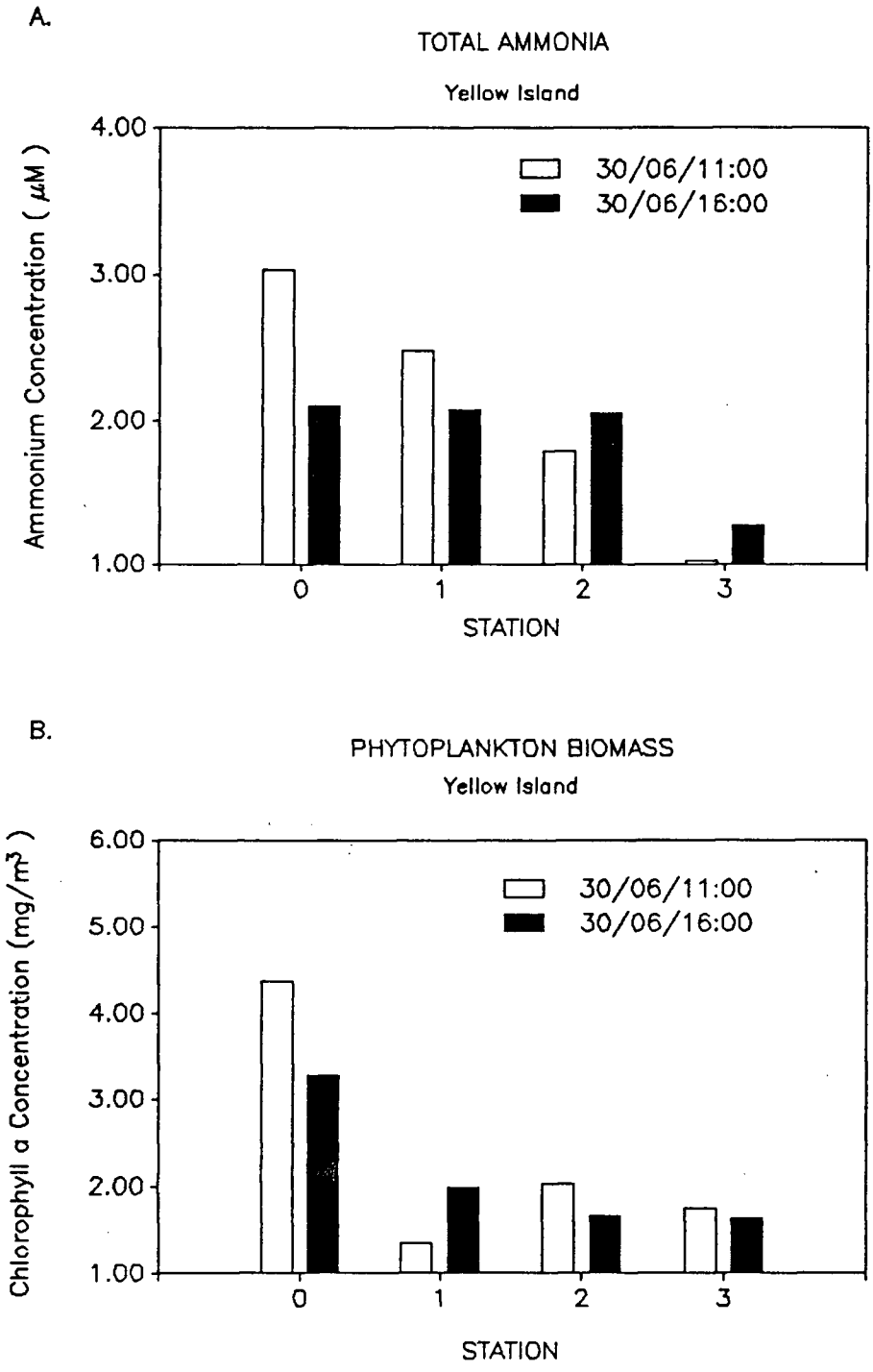


Figure 5. Total ammonia (A) and chlorophyll a (B) concentrations at Yellow Island, mid-August, 1988.

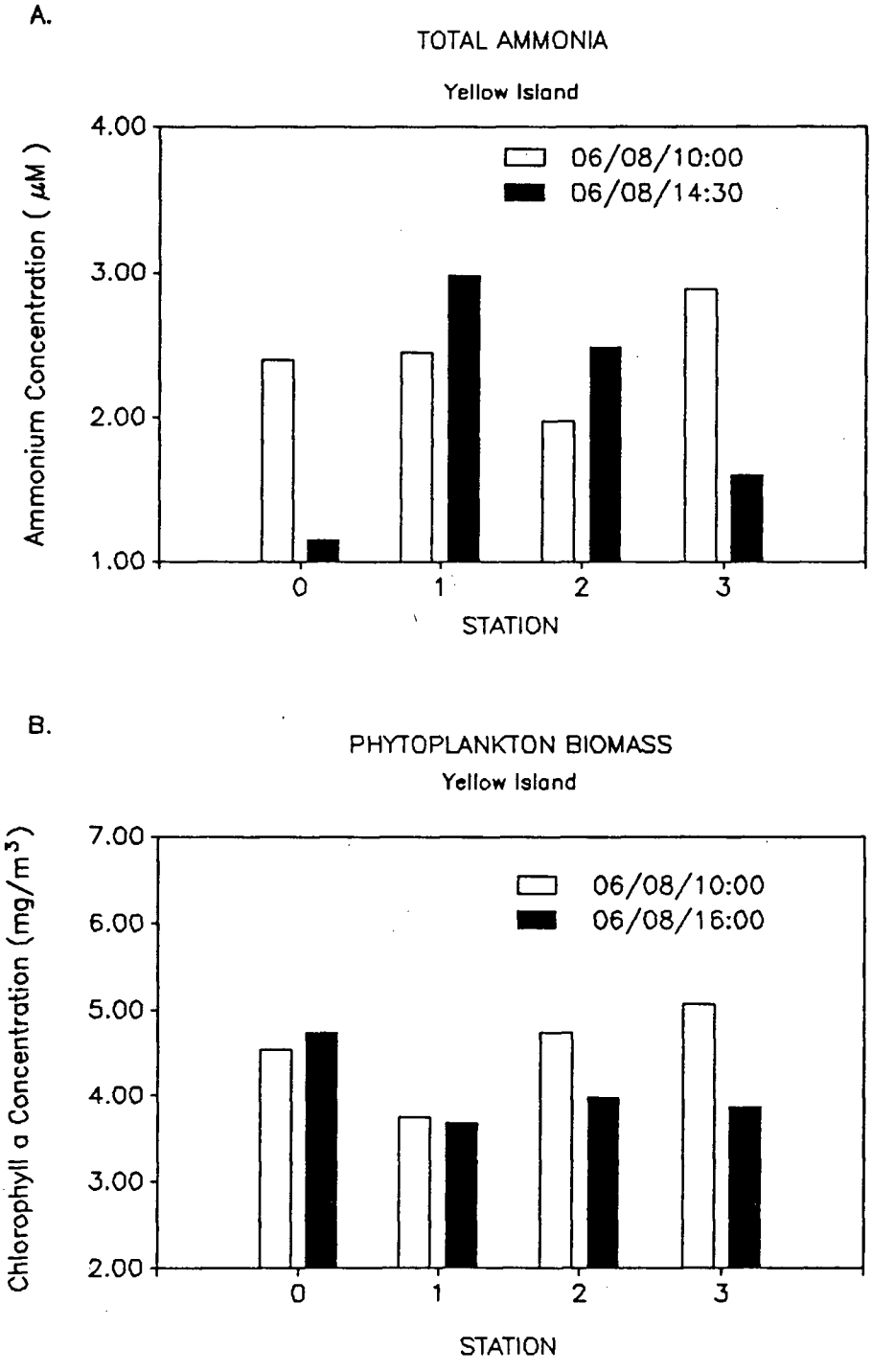


Figure 5. Con't.

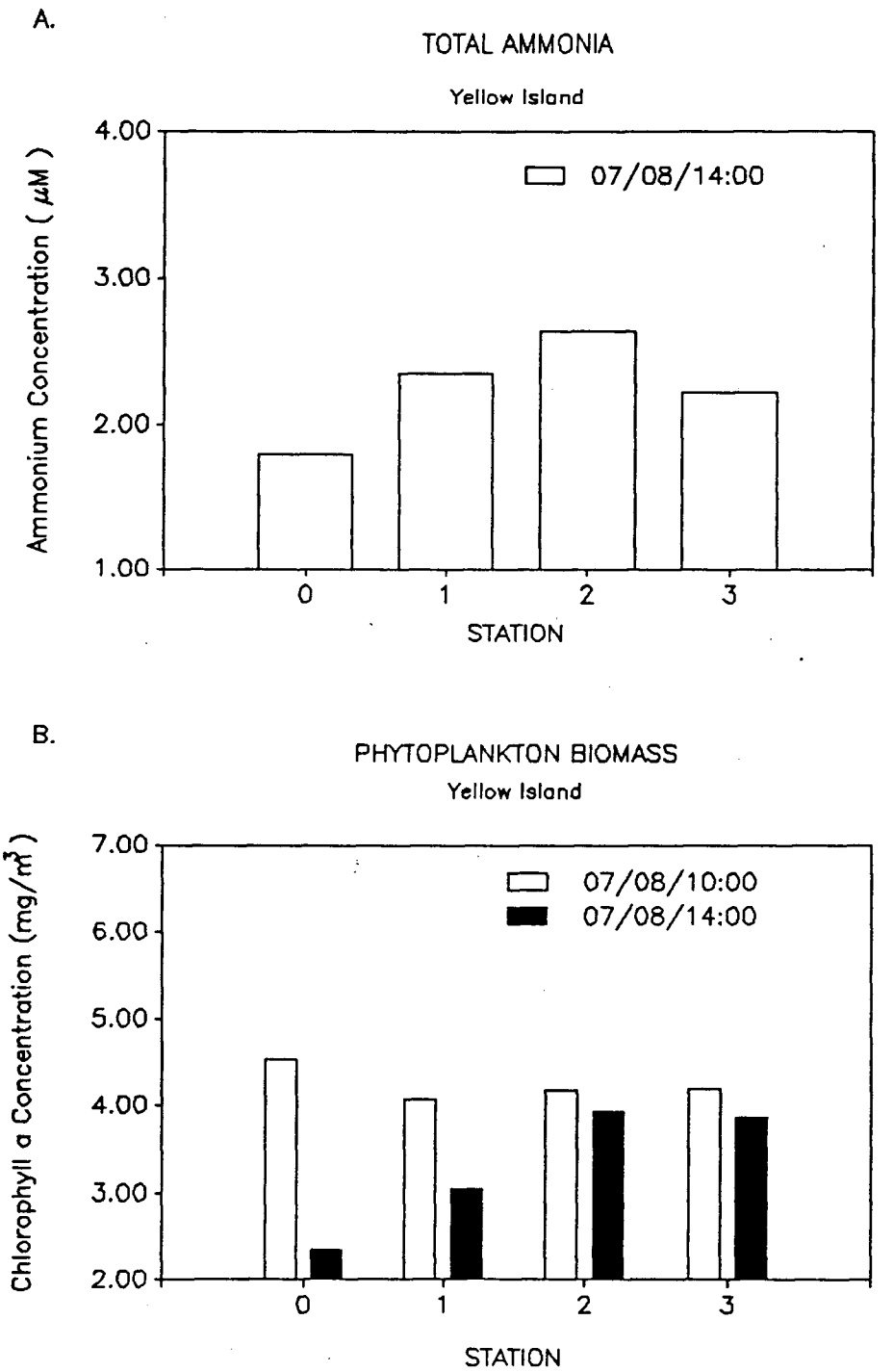


Figure 6. Total ammonia (A) and chlorophyll a (B) concentrations at Yellow Island, late September, 1988.

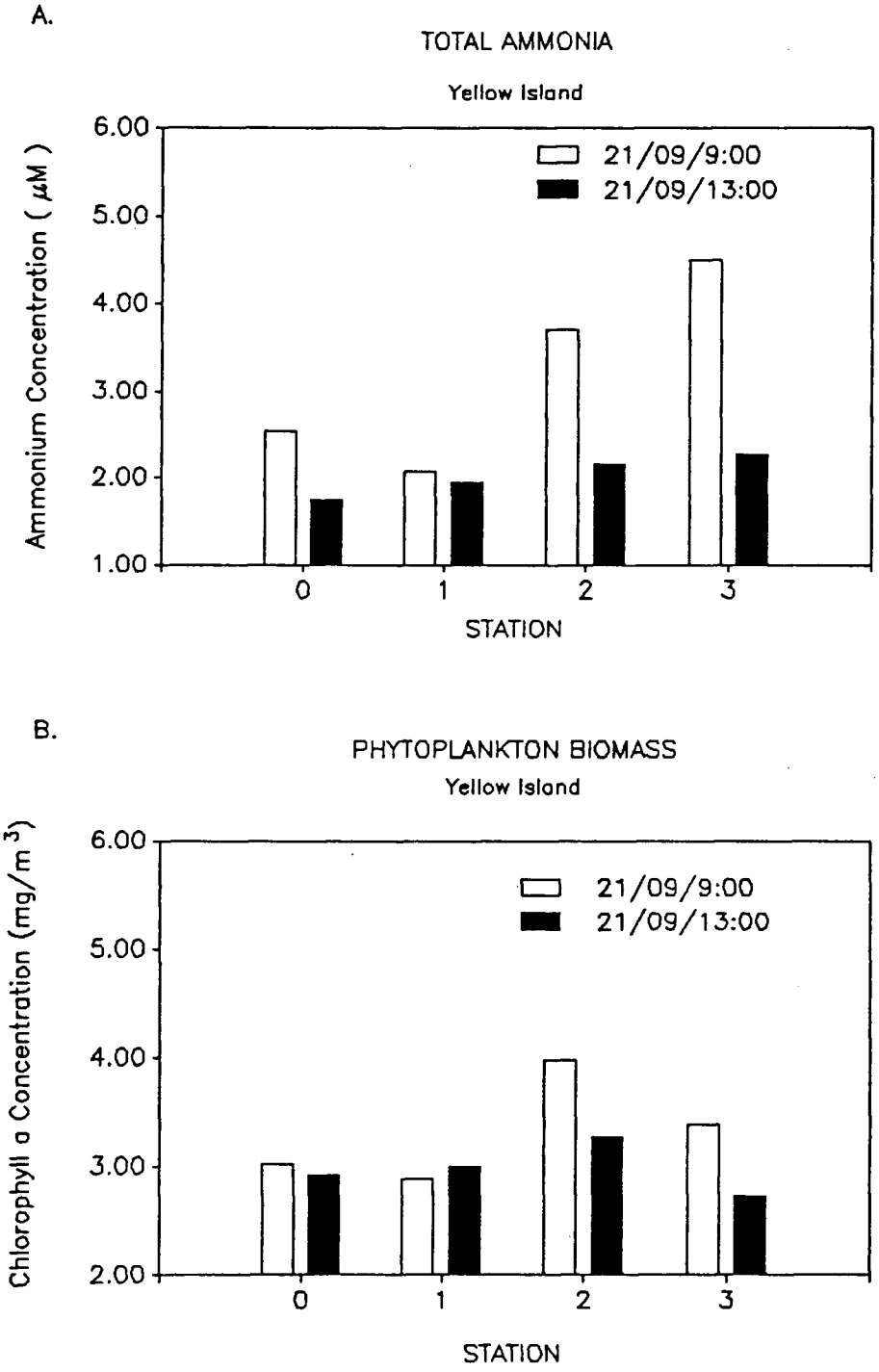


Figure 7. Total ammonia (A) and chlorophyll a (B) concentrations at Quartz Bay, summer, 1988.

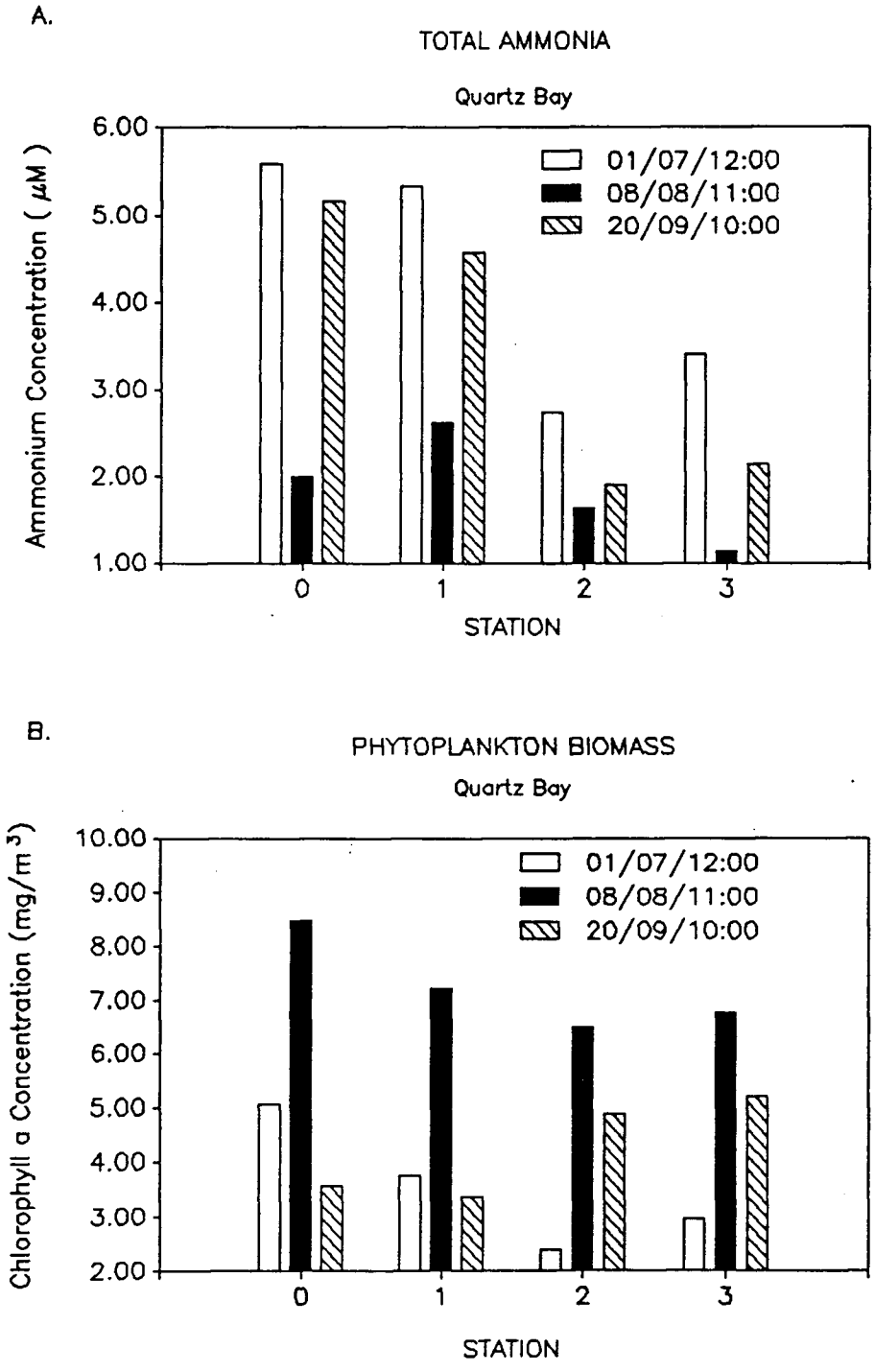
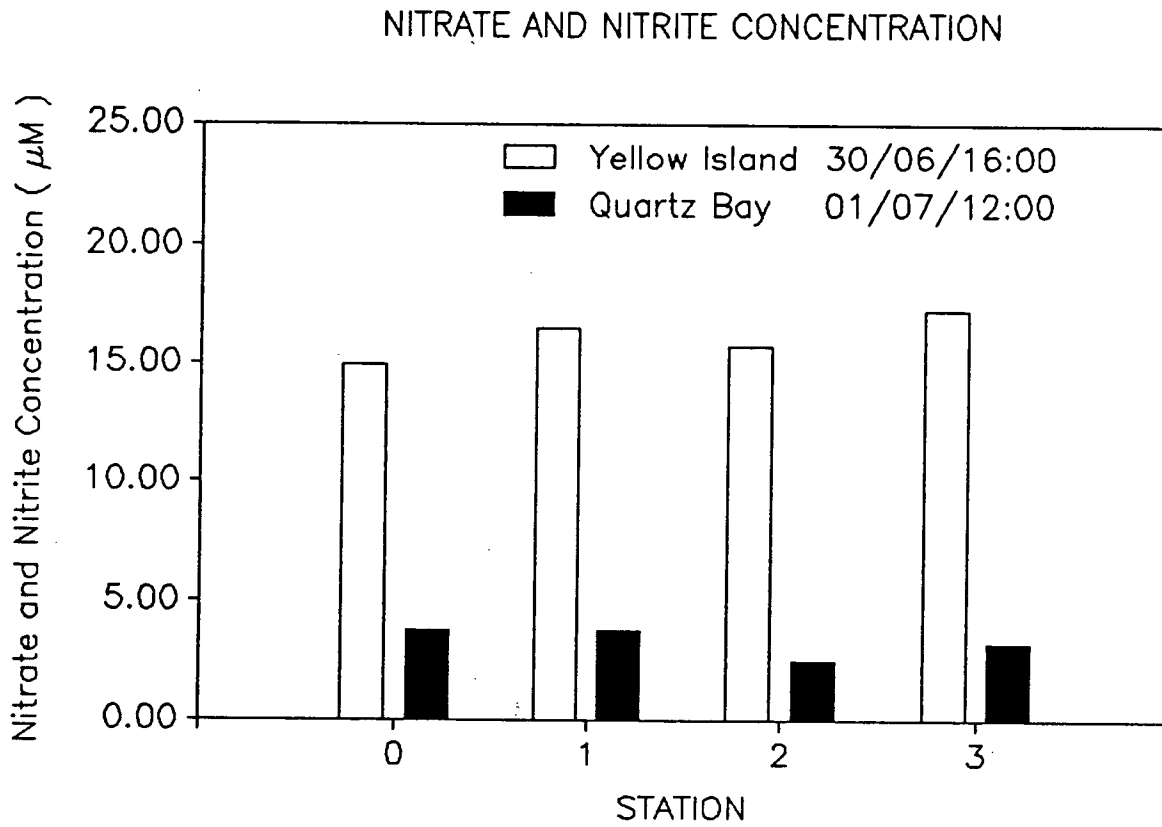


Figure 8. Nitrate and nitrite concentration at Yellow Island and Quartz Bay, late June/early July, 1988.



When data from all sampling periods were combined for each station, no significant differences ($p<.05$) were observed between all four stations for total ammonia concentrations at either Quartz Bay or Yellow Island (Table 7). If the inner and outer two stations are lumped together , however, a significantly higher total ammonia level ($p<.05$) in the immediate vicinity of the pens is seen at Quartz Bay (Table 8). The effects of freezing on the variability of the total ammonia determinations made in this study are discussed in appendix 1.

Acidic contamination resulted in the loss of all nitrate samples with the exception of those taken at the first sampling period at both Yellow Island and Quartz Bay. Nitrate values at both locations appeared very constant between stations (Fig. 8), with the mean value at Yellow Island (15.0 *u*M) considerably higher than the mean at Quartz Bay (2.5 *u*M).

**C. Phytoplankton Standing Stock**

Surface concentrations of chlorophyll a at Stations 0 and 3 at Yellow Island ranged from 2.35 to 4.73 mg Chl $a \cdot m^{-3}$ and 1.63 to 5.06 mg Chl $a \cdot m^{-3}$, respectively (Table 5). The only evidence of higher chlorophyll a levels in the immediate vicinity of the net-pens is seen in late June (Fig. 4B). All other sampling periods show no obvious patterns of chlorophyll a distribution between stations (Fig.'s 5B, and 6B).

Quartz Bay summer chlorophyll a surface concentrations ranged from 3.58 to 8.47 mg Chl $a \cdot m^{-3}$ at Station 0 and 3.00 to 6.77 mg Chl $a \cdot m^{-3}$ at Station 3 (Table 6). Noticeably higher concentrations of chlorophyll a in the immediate vicinity of the net-pens were observed in early July and mid-August (Fig. 7B). Surface chlorophyll a concentrations at both Yellow Island and Quartz Bay showed no significant differences ($p<.05$) between all stations (table 7) and between the inner and outer station groupings (table 8).

**D. Primary Productivity**

Inspection of figure 9A reveals that there was a high amount of variability in the rates of primary production at Yellow Island at all stations over the course of the summer. The range of photosynthetic rates were larger at Station 0 (2.67 to 8.23 mg $C \cdot m^{-3} \cdot hr^{-1}$) compared to those at Station 3 (0.88 to 4.66 mg $C \cdot m^{-3} \cdot hr^{-1}$). The ranges of production/biomass (P/B) ratios seen at Stations 0 and 3 (Table 5) do not show any marked difference. Although rates of primary production appear to be enhanced in the vicinity of the net-pens in June and August (Fig. 9A), this effect is not reflected in the corresponding productivity indices (Fig. 10A).

Primary productivity estimates at Quartz Bay also exhibited a large amount of variability over the course of the summer (Fig. 9B). Photosynthetic rates at Station 0 ranged from 5.16 to 14.12 mg $C \cdot m^{-3} \cdot hr^{-1}$ while those at

Station 3 varied from 4.33 to 10.61 mg $C \cdot m^{-3} \cdot hr^{-1}$. These production values are considerably higher than those observed at Yellow Island. Although the production/biomass ratios at Quartz Bay (Table 6) are on the whole higher than those for Yellow Island, the difference between the two locations P/B ratios is smaller than the difference in primary production values. Increased rates of primary production in the vicinity of the net-pens at Quartz Bay were observed only in August (Fig. 9B), however this effect was not seen in the pattern of productivity indices between stations at any point in the summer (Fig. 10B). No significant differences ($p<.05$) in photosynthetic rates or P/B ratios at Quartz Bay or Yellow Island were detected between all stations (Table 7) or between the inner and outer station groupings (Table 8).

Table 7. Summary of results of Kruskal-Wallis (K-W) One-way Analysis of Variance to determine significant differences between all stations at Yellow Island and Quartz Bay. (-) denotes an acceptance (PROB > .05) of the null hypothesis ($H_o$), demonstrating no significant differences between stations.

| WATER QUALITY PARAMETER | LOCATION | SAMPLE SIZE | K-W TEST STATISTIC | PROB. ( p ) | Ho . |
|---|---|---|---|---|---|
| TOTAL AMMONIA | Yellow | 28 | 0.771 | .856 | - |
| | Quartz | 12 | 4.385 | .223 | - |
| CHLOROPHYLL A | Yellow | 32 | 2.830 | .419 | - |
| | Quartz | 12 | 0.641 | .887 | - |
| PRIMARY PRODUCTION | Yellow | 16 | 1.346 | .718 | - |
| | Quartz | 12 | 0.282 | .963 | - |
| P/B | Yellow | 16 | 1.994 | .574 | - |
| | Quartz | 12 | 1.564 | .658 | - |
| BACTERIA | Yellow | 32 | 1.645 | .649 | - |
| | Quartz | 12 | 1.051 | .789 | - |
| DOC ABSORBANCE | Yellow | 32 | 6.108 | .106 | - |
| | Quartz | 12 | 2.179 | .536 | - |

Table 8. Summary of results of the Mann-Whitney (M-W) U-Test to determine significant differences between the means of the inner (stn. 0 and 1) and outer (stn. 2 and 3) stations for Yellow Island and Quartz Bay. (-) denotes an acceptance (PROB > .05) of the null hypothesis ($H_o$), demonstrating no significant differences between the inner and outer stations. (+) denotes rejection of the null hypothesis (PROB < .05).

| WATER QUALITY PARAMETER | LOCATION | SAMPLE SIZE | M-W TEST STATISTIC | PROB. ( p ) | $H_o$ |
|---|---|---|---|---|---|
| TOTAL AMMONIA | Yellow | 28 | 100.0 | .927 | - |
| | Quartz | 12 | 31.0 | .037 | + |
| CHLOROPHYLL A | Yellow | 32 | 120.0 | .763 | - |
| | Quartz | 12 | 21.0 | .631 | - |
| PRIMARY PRODUCTION | Yellow | 16 | 43.0 | .248 | - |
| | Quartz | 12 | 19.0 | .873 | - |
| P/B | Yellow | 16 | 43.5 | .227 | - |
| | Quartz | 12 | 21.0 | .631 | - |
| BACTERIA | Yellow | 32 | 94.0 | .200 | - |
| | Quartz | 12 | 12.0 | .337 | - |
| DOC ABSORBANCE | Yellow | 32 | 63.5 | .015 | + |
| | Quartz | 12 | 27.0 | .150 | - |

Figure 9. Hourly and daily photosynthetic rates at Yellow Island (A) and Quartz Bay (B), summer, 1988. Bars indicate 1 standard error of the mean.

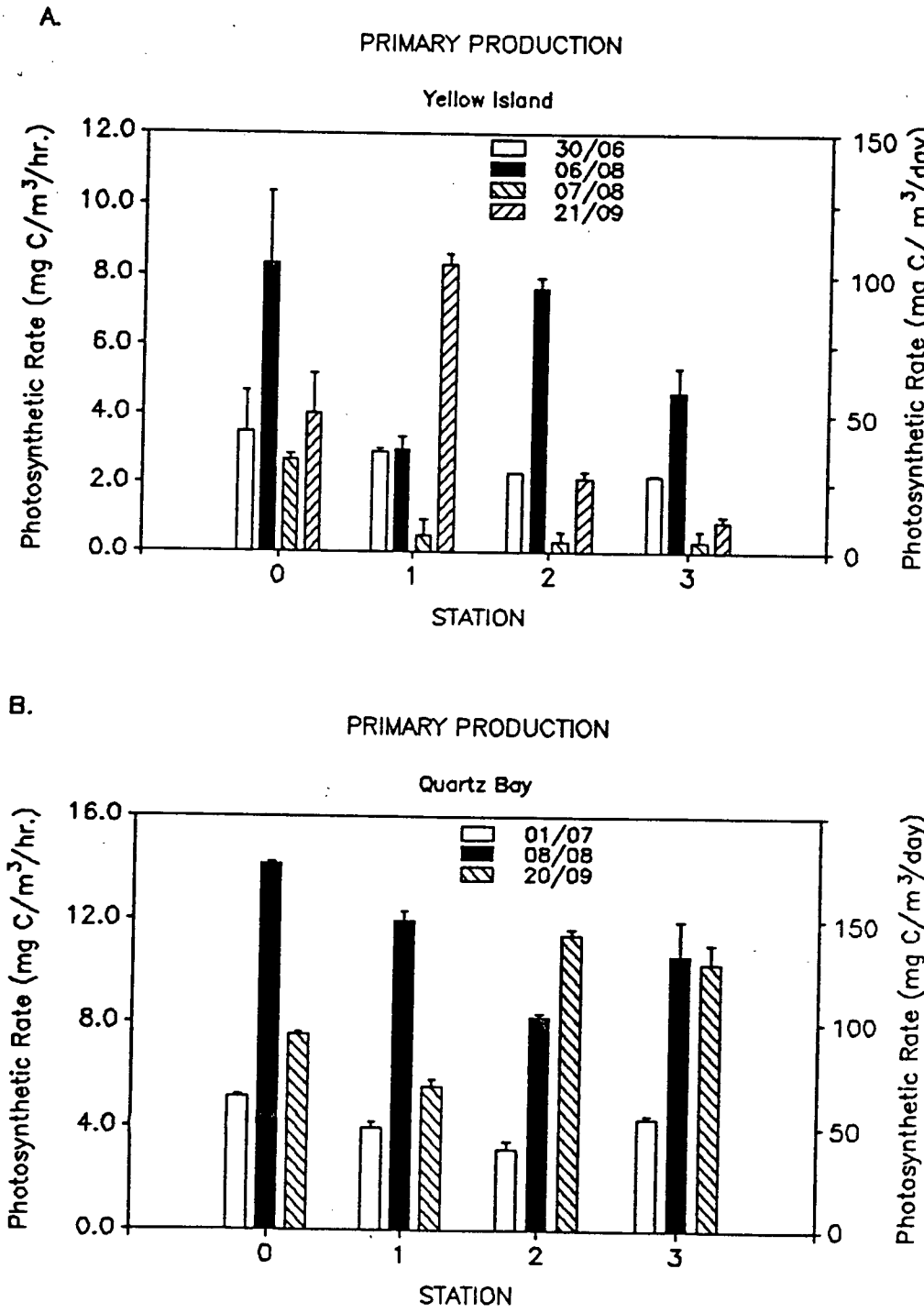
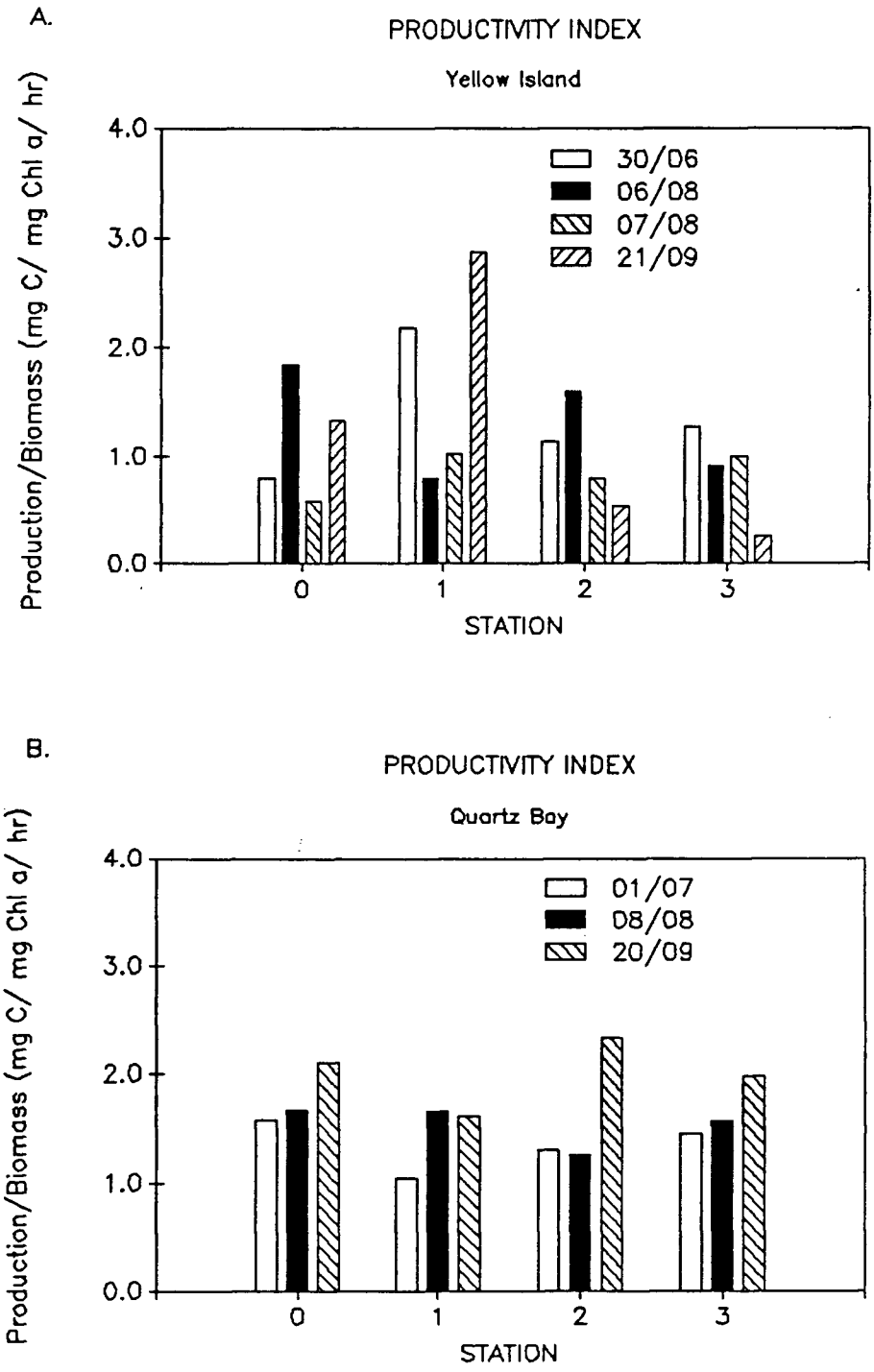


Figure 10. Productivity indices (production/biomass) at Yellow Island (A) and Quartz Bay (B), summer, 1988.



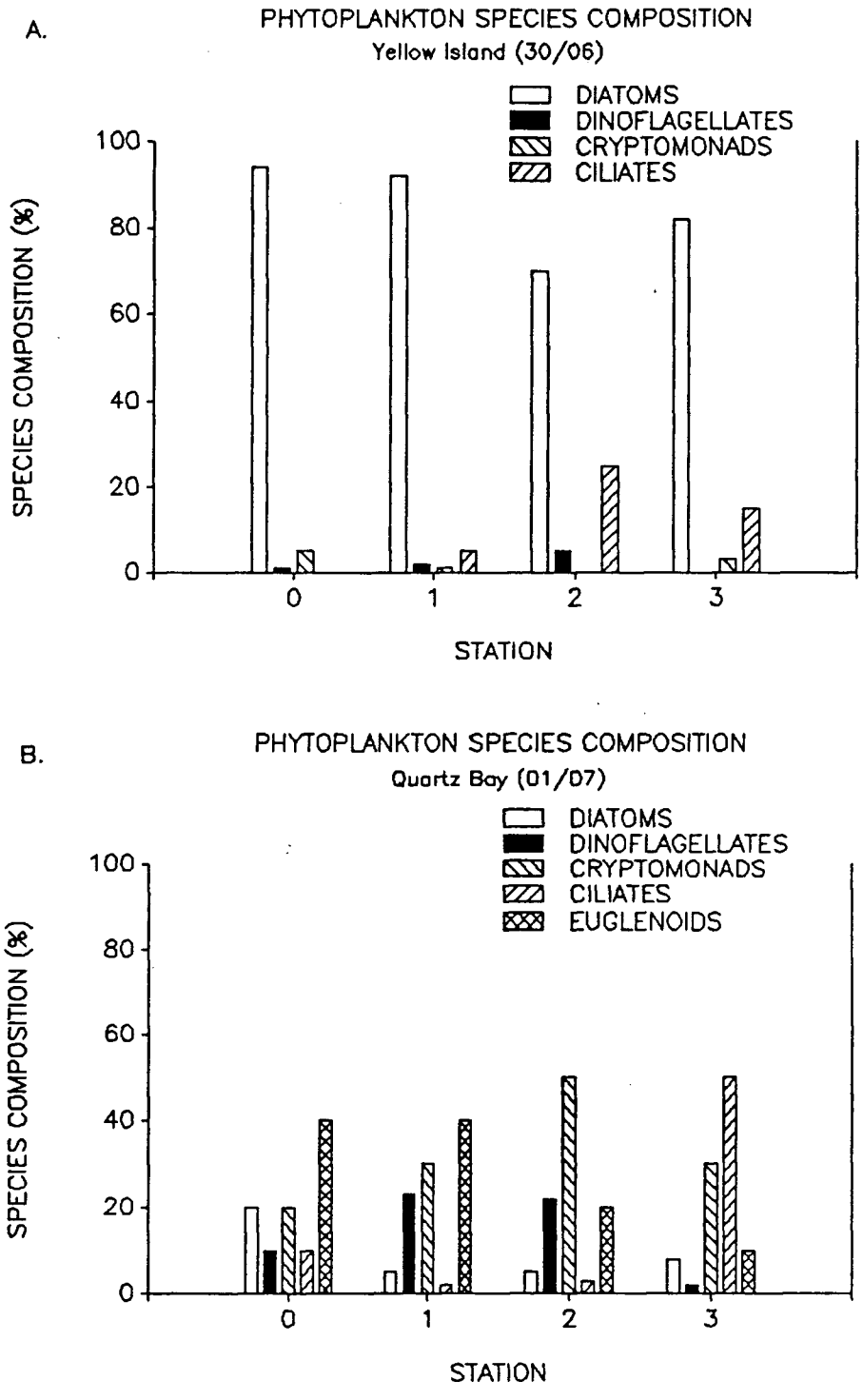
**E. Phytoplankton Species Composition**

Diatoms were the dominant class of phytoplankton observed at Yellow Island throughout the course of the summer. A number of benthic diatoms including, *Melosira moniliformis*, *Fragilaria* spp., naviculoids and *Odontella aurita* as well as the pelagic species *Skeletonema costatum* were the dominant organisms observed in late June and comprised 70 to 95% of the total phytoplankton biomass (Fig. 11A). Low levels of dinoflagellates and cryptomonads were only seen in late June at which time the photosynthetic ciliate *Mesodinium rubrum* reached dominance levels of up to 25% at the outer two stations (Fig. 11A). By the second sampling period (August 6), the planktonic diatoms *Skeletonema costatum*, *Nitzschia pungens*, *Chaetoceros debile* and *Thalassiosira rotula* comprised 95 to 98% of the biomass of phytoplankton and this pattern continued into the final sampling period in September.

The species composition at Quartz Bay during the first sampling period (July 1) exhibited a much more varied composition than that of Yellow Island. Maximum diatom dominance in early July was observed at Station 0 (20%) and dropped to values as low as 5% at the outer stations. This diatom biomass was made up mainly of *Odontella aurita*, *Melosira moniliformis* and *Eucampia zoodiacus*. A much larger proportion of the Quartz Bay biomass was comprised of heterotrophic dinoflagellates (20%), cryptomonads (30%) and

euglenoids (40%). *Mesodinium rubrum* reached maximum abundance levels of 50% of the total at the outer stations (Fig. 11B). By the second sampling period (August 8), the diverse species composition had been replaced by one consisting of 95% diatoms. The diatoms *Skeletonema costatum*, *Nitzschia pungens*, *Thalassiosira aestivalis* and *Thalassiosira rotula* remained dominant until the final sampling period in late September. A complete list of the phytoplankton species observed at Yellow Island and Quartz Bay over the course of the summer is given in Table 12, Appendix II.

Figure 11. Phytoplankton species composition at Yellow Island (A) and Quartz Bay (B), late June/ early July, 1988.



## F. Dissolved Organic Carbon Absorbance and Bacterial Numbers

Dissolved organic carbon absorbance (*i.e.* 280 nm absorbance) at Yellow Island and Quartz Bay ranged from .090 to .253 units and .105 to .252 units, respectively (Tables 5 and 6). Figures 12A through 15A do not demonstrate any DOC enrichment in the vicinity of the net-pens. No significant differences ($p<.05$) in DOC absorbance between all stations (Table 7) were found for either Yellow Island or Quartz Bay. Significantly higher DOC absorbances ($p<.05$) were observed at the outer stations at Yellow Island but not at Quartz Bay (Table 8).

Bacterial concentrations at Yellow Island ranged from 3.36 to 9.92 x $10^5$ cells·ml$^{-1}$ (Table 5). Bacterial numbers at Quartz Bay were on average two-fold higher with concentrations ranging from 5.20 to 18.40 x $10^5$ cells·ml$^{-1}$ (Table 6). Inspection of Figures 12B through 15B reveals no obvious patterns in the spatial distributions of bacteria at both locations. Bacterial concentrations do appear to drop off during the last sampling periods at both Yellow Island and Quartz Bay. No significant differences ($p<.05$) in bacterial numbers between all stations (Table 7) and between the inner and outer station groupings (Table 8) were found for either Yellow Island or Quartz Bay.

Figure 12. DOC absorbance (A) and bacterial numbers (B) at Yellow Island, late June, 1988. Bars indicate 1 standard error of the mean.

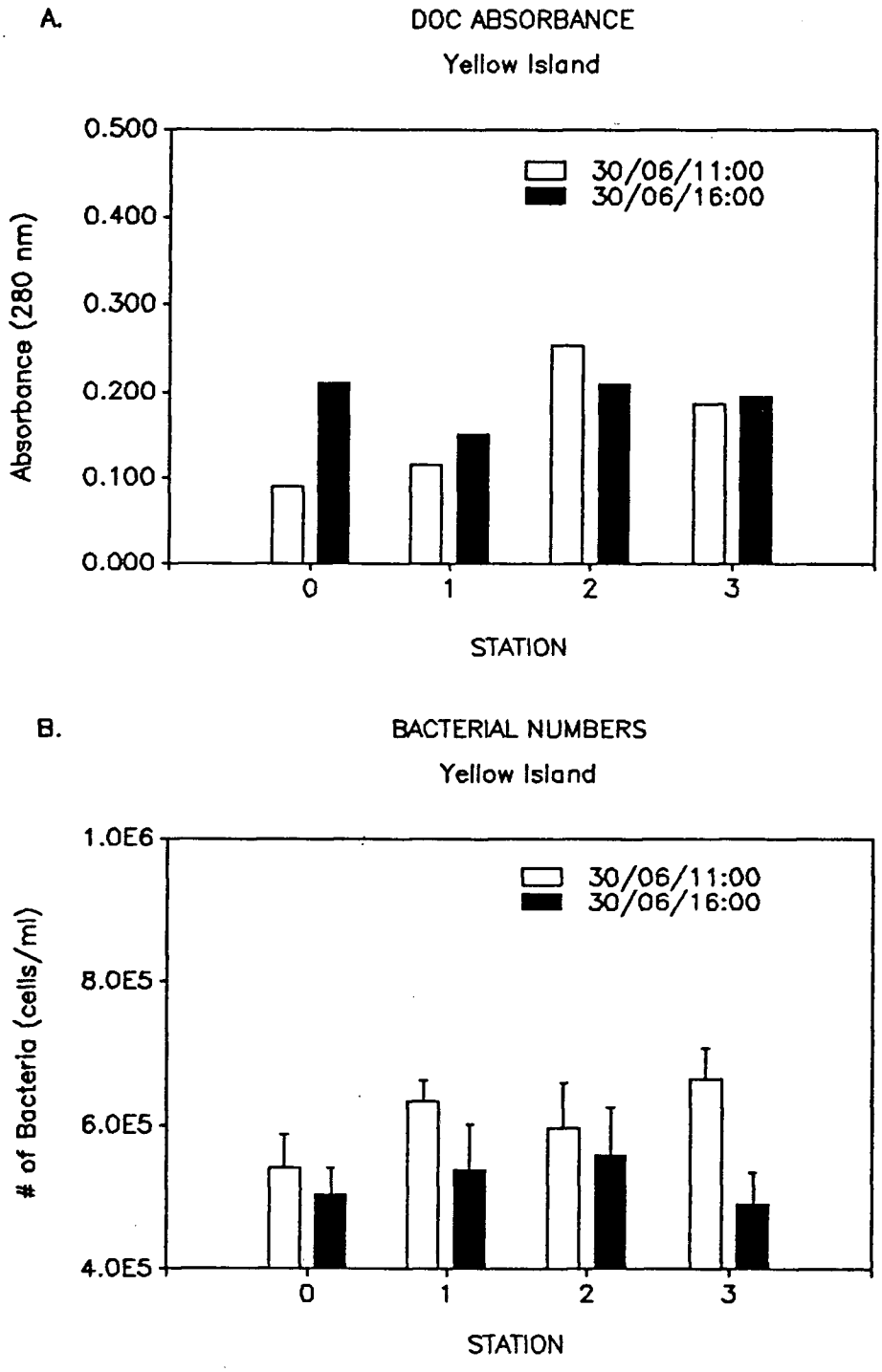


Figure 13. DOC absorbance (A) and bacterial numbers (B) at Yellow Island, mid-August, 1988. Bars indicate 1 standard error of the mean.

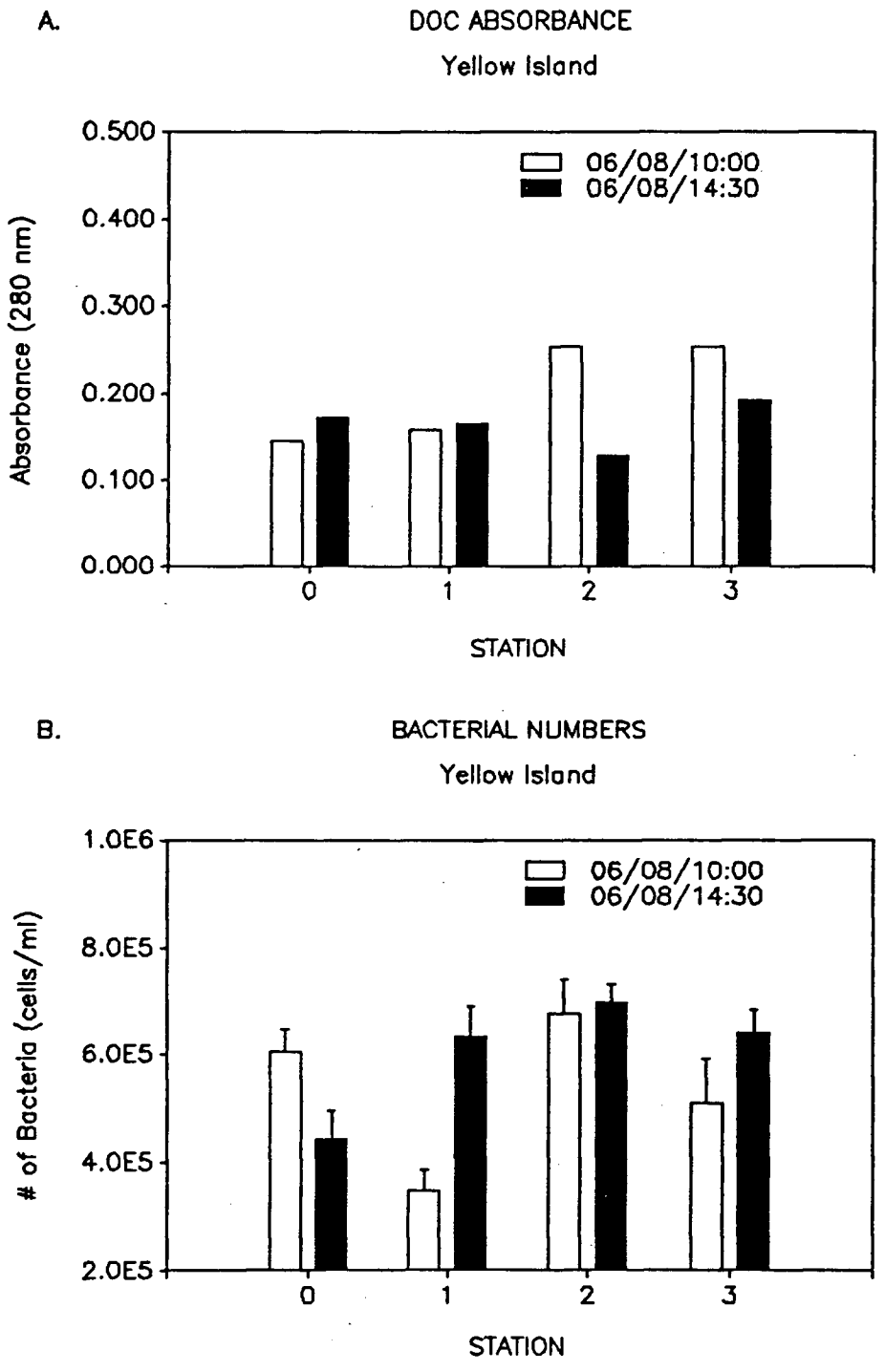


Figure 13. Con't.

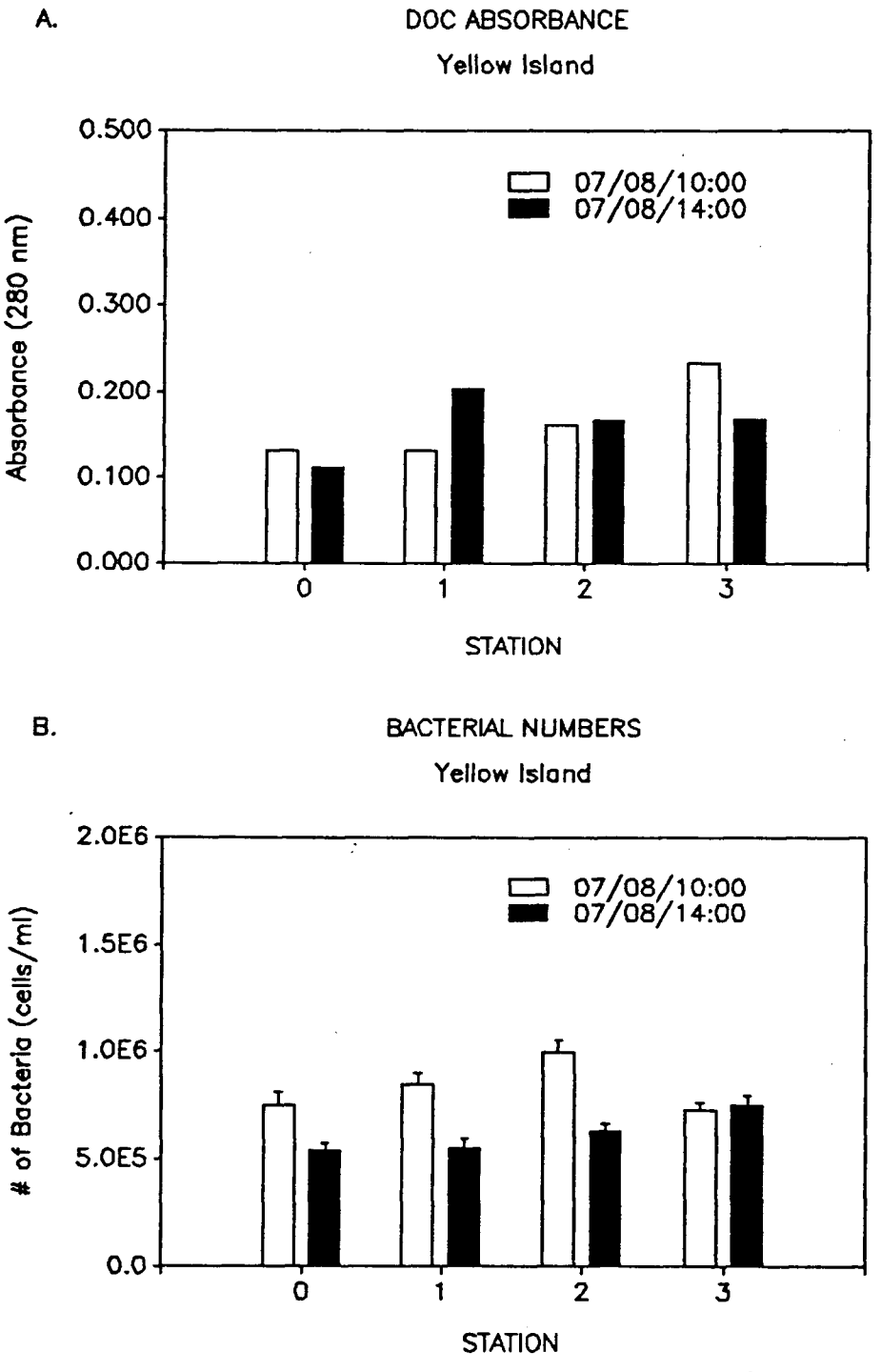


Figure 14. DOC absorbance (A) and bacterial numbers (B) at Yellow Island, late September, 1988. Bars indicate 1 standard error of the mean.

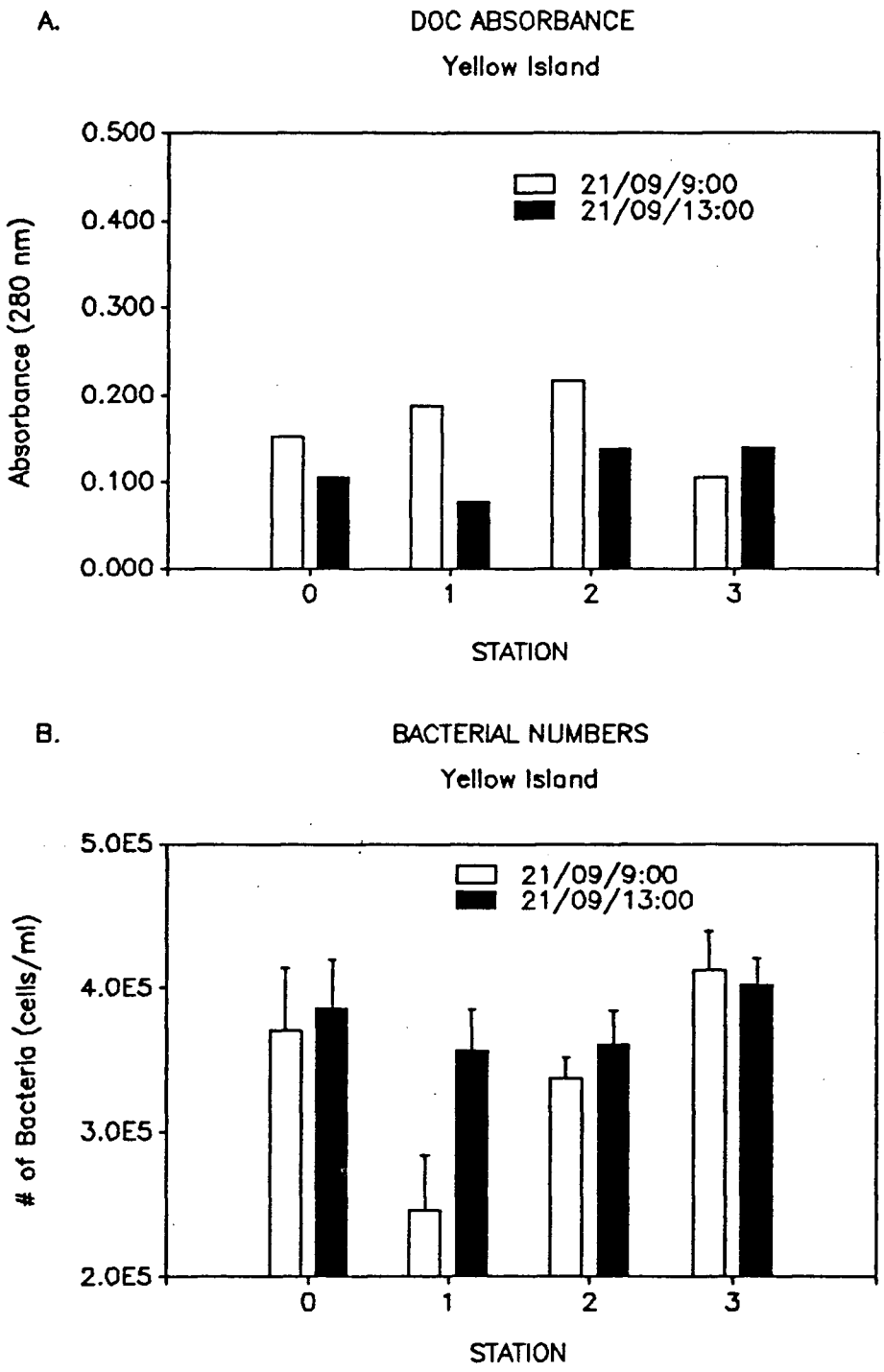
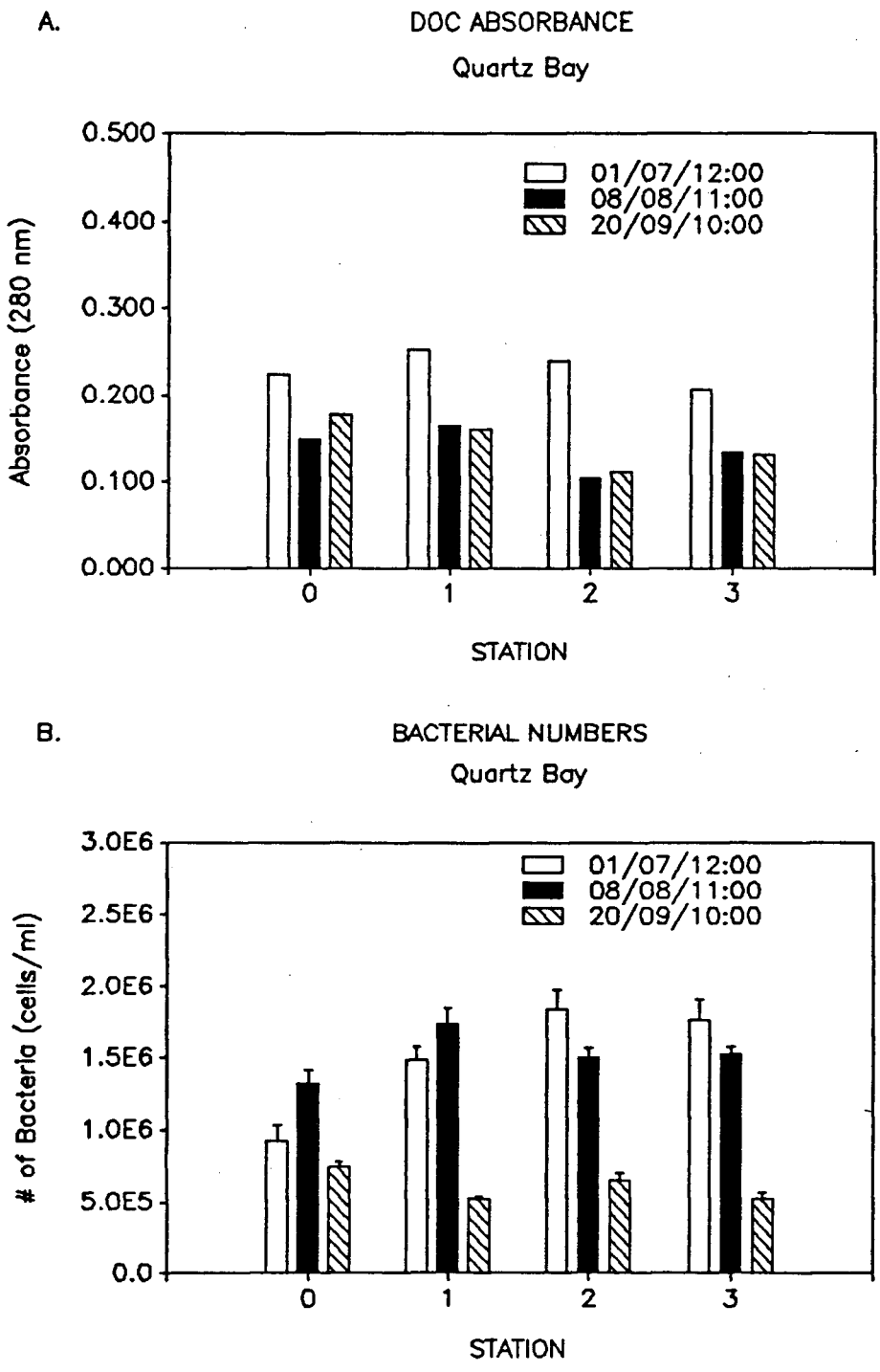


Figure 15. DOC absorbance (A) and bacterial numbers (B) at Quartz Bay, summer, 1988. Bars indicate 1 standard error of the mean.



## G. Water Quality Parameter Interactions

Summaries of the results of the Spearman correlation matrices for the Yellow Island and Quartz Bay water quality data are given in Tables 9 and 10, respectively. To construct these matrices, data collected over the entire summer at each station were lumped together for both Quartz Bay and Yellow Island. The correlations are therefore useful in determining the effects of seasonal changes in physical parameters on the various water quality parameters investigated. The correlation between chlorophyll a and rates of photosynthesis at Quartz Bay demonstrates the large degree of influence that phytoplankton standing stock has on estimates of primary productivity and warrants the use of P/B ratios to estimate true changes in carbon uptake rates. Many of the biological parameters at Quartz Bay are correlated with seasonal changes in physical variables such as water column stratification and surface temperature. Chlorophyll a and primary productivity are positively correlated with surface temperature at Yellow Island and water column stratification at Quartz Bay. A summary of the effect of the salmon farms on the water quality measures investigated in this study is given in Table 11.

Table 9. Spearman nonparametric correlation matrix of the Yellow Island water quality data Sample size for each coefficient is 12.

| | AMM. | CHL A. | C14. | PB. | BAC. | DOC. |
|---|---|---|---|---|---|---|
| **AMM.** | | | | | | |
| **CHL A.** | | | | | | |
| **C14.** | | | | + | | |
| **PB.** | - | | | | | |
| **BAC.** | | | | | | |
| **DOC.** | | | | | | |
| **STRAT.** | | | | | - | |
| **TEMP.** | | ++ | + | | | |
| **SALINITY** | | | | | - | |
| **SECCHI.** | | -- | | | - | |

+ = Significant positive correlation ( p < .05 ).
++ = Significant positive correlation ( p < .01 ).
- = Significant negative correlation ( p < .05 ).
-- = Significant negative correlation ( p < .01 ).

AMM. = Total Ammonia concentration
CHL A. = Chlorophyll a concentration
C14. = Primary Productivity
PB. = Production/Biomass ratio
DOC. = Dissolved Organic Carbon absorbance
BAC. = Bacterial numbers
STRAT. = Stratification
TEMP. = Surface Temperature
SALINITY= Surface Salinity
SECCHI = Secchi Disc Depth

Table 10. Spearman nonparametric correlation matrix for the Quartz Bay water quality data. Sample size for each coefficient is 12.

| | AMM. | CHL A. | C14. | PB. | BAC. | DOC. | . |
|---|---|---|---|---|---|---|---|
| **AMM.** | | | | | | | |
| **CHL A.** | - | | | | | | |
| **C14.** | - | ++ | | + | | | |
| **PB.** | | | | | | | |
| **BAC.** | | | | - | | | |
| **DOC.** | ++ | - | -- | | | | |
| **STRAT.** | -- | ++ | ++ | | | -- | |
| **TEMP.** | | | - | -- | ++ | + | |
| **SALINITY** | | - | | | - | | |
| **SECCHI.** | | | | | - | | |

\+ = Significant positive correlation ( $p < .05$ ).
++ = Significant positive correlation ( $p < .01$ ).
\- = Significant negative correlation ( $p < .05$ ).
-- = Significant negative correlation ( $p < .01$ ).

AMM. = Total Ammonia concentration
CHL A. = Chlorophyll a concentration
C14. = Primary Productivity
PB. = Production/Biomass ratio
BAC. = Bacterial numbers
DOC. = Dissolved Organic Carbon absorbance
STRAT. = Stratification
TEMP. = Surface Temperature
SALINITY= Surface Salinity
SECCHI = Secchi Disc Depth

Table 11. Summary of changes in physical and biological water quality parameters at Yellow Island and Quartz Bay over the course of the summer, 1988. (++) denotes the presence of a positive gradient in total ammonia or production measures from Station 0 or 1 to Station 3.

| PARAMETER | JULY | | AUGUST | | SEPTEMBER | |
|---|---|---|---|---|---|---|
| | YELLOW ISLAND | QUARTZ BAY | YELLOW ISLAND | QUARTZ BAY | YELLOW ISLAND | QUARTZ BAY |
| TOTAL AMMONIA | ++ | ++ | | | | ++ |
| CHLOROPHYLL A | ++ | ++ | | ++ | | |
| PHOTOSYNTHETIC RATE | ++ | | ++ | ++ | | |
| PRODUCTIVITY INDEX | | | | | | |
| DISSOLVED ORGANIC CARBON | | | | | | |
| BACTERIAL NUMBERS | | | | | | |
| STRATIFICATION | none | weak | none | strong | none | weak |
| SECCHI DISC DEPTH (m) | 11.0 | 9.0 | 7.2 | 5.0 | 10.0 | 5.5 |
| SURFACE WATER TEMPERATURE C | 10.6 | 15.8 | 11.2 | 14.3 | 10.7 | 10.9 |
| SURFACE SALINITY $^{o}/_{oo}$ | 29.2 | 24.1 | 29.3 | 22.9 | 29.7 | 24.9 |

# V. DISCUSSION

The physical measurements taken in this study indicate some marked differences in the flushing characteristics of the Yellow Island and Quartz Bay locations. Yellow Island experiences greater mixing as witnessed by the lack of any appreciable pycnocline compared to the strong stratification seen at Quartz Bay (Fig. 3). Maximum current speeds at Quartz Bay were considerably less than those measured at Yellow Island (Tables 3 and 4). Given the narrower width of the mouth of Quartz Bay and the more protected position of the farm within the embayment (Fig. 2), there is no doubt that the Yellow Island salmon farm undergoes a stronger flushing regime.

The timing and extent of total ammonia enrichment at the two locations vary considerably. The only strong evidence for higher ammonium levels in the immediate area of the farm at Yellow Island is seen in late June during the low tide sampling period (Fig. 4A). An understanding of the seasonal changes in ammonium concentrations in the Discovery Passage area may explain the timing of the Yellow Island enrichment. From work done on the Campbell River estuary, Seki *et al.* (1987 and 1984) demonstrated an increase in near-shore surface ammonium concentrations (*ca.* 10 km south of Yellow Island on western shore of Quadra Island) of 1.6 uM to 6.3 uM between May and August, 1983. This summertime increase followed a large rise in surface

salinity which reflects the intrusion of nutrient-rich coastal water and replacement of the less saline, nutrient-impoverished water associated with the spring freshet. If the rise in surface salinity and total ammonia levels in the outer two stations seen at Yellow Island over the course of the summer (Tables 3 and 5) are any indication of the trend in ambient ammonium levels, then one would expect to see the most noticeable total ammonia enrichment in the vicinity of the farm at the beginning of the summer when ambient concentrations were lowest. Rising background levels of ammonium over the course of the summer would swamp any enhancement in total ammonia concentrations resulting from the culture activities. The loss of the striking total ammonia gradient during the high tide sampling period in late June (Fig. 4A) could have been caused by the dilution of dissolved ammonium around the farm by the relatively ammonium-poor waters rising during the development of the near-spring high tide. Increases in ammonium uptake rates of phytoplankton during the latter part of the day resulting from exposure to higher light levels could also have contributed to decreases in total ammonia levels observed at the inner stations during the afternoon sampling period.

The spatial distribution of surface ammonium concentrations at Quartz Bay shows significant variation over the course of the summer (Fig. 7A). Conspicuous ammonium gradients between Stations 0 and 3 are seen in

early July and late September. During the August sampling period, however, total ammonia levels were considerably lower (Table 6) and no enrichment from the farm was apparent. During this time period, chlorophyll a concentrations were roughly two-fold higher than levels measured in early July and late September (Table 6). The lack of any ammonia enrichment may be related to the strong stratification observed in August (Fig. 3B) and its subsequent influence on phytoplankton biomass.

Stratification in the top 10 m in August at Quartz Bay was over three fold greater than in early July and two fold greater compared to late September (Table 4). The strong correlation ($P<.01$) between stratification and chlorophyll a (Table 10) suggests that increasing stability of the water column at Quartz Bay leads to higher levels of phytoplankton biomass. Elevations in algal ammonium uptake and utilization per volume of water, would of course, accompany this rise in biomass. Thus it is not surprising that total ammonia levels at Quartz Bay are severely depressed during periods when chlorophyll a levels are very high (Fig.'s 7A and 7B). The strong negative correlation ($p<.01$) between chlorophyll a and total ammonia confirms this observation (Table 10). The lack of any enhancement of total ammonia levels in the vicinity of the farm during the August sampling period is surely the result of increases in algal ammonium uptake per volume of water during this time period.

The total ammonia data presented here also give some indication of the importance of ammonium loading caused by the excretion of fouling organisms. Total ammonia levels within the net-pens at Yellow Island and Quartz Bay during periods of maximum enrichment were respectively 2.0 and 2.5 fold higher than ambient levels measured at Station 3. As outlined in the introduction, the salmon farms that were investigated in this study would produce roughly 20 $kg \cdot NH_4\text{-}N \cdot day^{-1}$ through salmonid excretion alone. Under stagnant flushing conditions, this loading would produce an increase of *ca.* 100 *u*M $NH_4\text{-}N \cdot day^{-1}$ within the pens assuming a total pen volume of 12000 $m^{-3}$ (12 stocked pens with volumes of 1000 $m^{-3}$). From the current speeds measured immediately downcurrent of the net-pens (Tables 3 and 4), the minimum flushing time for any pen based on the lowest current speed observed (4.0 $cm \cdot s^{-1}$) would be just over 4 minutes (*ca.* 350 $volumes \cdot day^{-1}$). Considering this flushing rate, the maximum increase in total ammonia levels inside the pens should be no greater than 0.30 *u*M $NH_4\text{-}N$ at any time (100 *u*M $NH_4\text{-}N \cdot day^{-1}$/350 $flushings \cdot day^{-1}$). The elevated total ammonia levels seen within the pens at Quartz Bay and Yellow Island must therefore be a result of more than ammonium loading from salmonid excretion alone. The only other significant source of ammonium in the pens would come from the excretion of fouling organisms on the nets and raft structures.

The effect of the elevated ammonium levels on the photosynthetic rates and densities of phytoplankton surrounding the culture facilities will depend on the ambient light, temperature and nutrient conditions to which the algal cells are exposed. The compensation depths calculated for Yellow Island and Quartz Bay ranged from 13-30 m (Tables 3 and 4). Given the high light levels and long days characteristic of the summer months, the deep compensation depths observed at Yellow Island and the strong stratification seen at Quartz Bay, it seems unlikely that phytoplankton found at the surface would be light limited at either location during the summer. Temperature and ambient nitrogen levels at the two locations were markedly different and require separate discussion.

The positive correlation ($p<.05$) between temperature and primary production at Yellow Island (Table 9) suggests that phytoplankton production may have been temperature-limited in this area. Low surface temperatures never exceeding 11.6 C over the course of the summer reinforce this hypothesis (Table 3). If this information is coupled with the high nitrate concentrations (15 uM) measured in late June at Yellow Island (Fig. 8) during a time when they should be at a seasonal low (Seki *et al.*, 1987 and 1984), it would seem unlikely that any ammonium enrichment associated with the farm would result in higher levels of primary production. The increased photosynthetic rates observed during late June and mid-August (Fig. 9A) in the vicinity of

the pens do not support this hypothesis. When the true carbon uptake is derived by dividing photosynthetic rates by chlorophyll a levels, however, no spatial trends in primary production (i.e. productivity indices) are observed (Fig. 10A). As we shall see later, the lack of any enhancement in carbon uptake rates in the vicinity of the farms may also have been related to methodological time scale errors.

Temperature and nitrate conditions were considerably different at Quartz Bay compared to Yellow Island. Surface temperatures at Quartz Bay over the course of the summer (Table 4) were much higher than those observed at Yellow Island. No positive correlations between temperature and photosynthetic rates or productivity indices were noted (Table 10) suggesting that temperature was not a growth-limiting factor at Quartz Bay. Coupled with the low nitrate levels at all stations (2.0-3.0 *u*M) observed in early July (Fig. 8), an enhancement of primary production in the vicinity of the farm at Quartz Bay would be possible. A noticeable increase in photosynthetic rates at all stations and higher rates at the inner stations were observed during the August sampling period (Fig. 9B). The higher photosynthetic rates undoubtedly resulted from elevated chlorophyll a levels during this period as proven by the strong correlation ($p<.01$) between these two variables at Quartz Bay (Table 10). The productivity indices which are a more accurate reflection of true cellular carbon uptake

rates showed no spatial trends between stations during this time period (Fig 10B).

Increased chlorophyll a concentrations at the inner two stations were observed in late June at Yellow Island (Fig. 4B) and early July and mid-August at Quartz Bay (Fig. 7B). Oddly, no increases in the primary productivity indices at these stations were noted during these sampling periods (Fig. 10). The explanation for this paradox lies in the difficulty of interpreting *in situ* measurements of carbon uptake with regard to enhanced community growth rates or elevated phytoplankton biomass. According to Harris (1988), the lack of any increase in carbon uptake following the exposure of phytoplankton to pulses of dissolved inorganic nitrogen (DIN) simply reflects the present state of the internal storage pool of DIN. Following exposure to high levels of ammonium, cells with depleted internal DIN pools would not be expected to show increases in carbon uptake until many hours after the exposure because the time scale for nutrient uptake and metabolism (i.e. carbon uptake) is in the order of hours. The time scale for cells to be transported from Station 0 to Station 3 is, however, in the order of minutes. It would be unreasonable, therefore, to expect to see a decrease in carbon uptake at Station 3 compared to the rate measured at Station 0 within minutes after the cells had been exposed to high levels of total ammonia. The replete internal storage pools of DIN within the cells would be sufficient to maintain high levels

of carbon uptake for many hours (Harris, 1988). In order to detect enhanced photosynthetic rates resulting from fish farm ammonium loading, one would have to measure *in situ* carbon uptake before and after the exposure of phytoplankton to the elevated ammonium levels associated with the culture facility. The paradox between the enhanced levels of chlorophyll a and constant P/B ratios is a result of employing too short a time scale to measure changes in metabolic processes which ultimately lead to changes in phytoplankton biomass.

Because the degree of correlation between nitrogen limitation and P/B ratios in natural systems is often poor (Turpin, 1983), it would be very difficult to determine if elevated phytoplankton biomass levels in the vicinity of a culture operation are resulting from high ammonium concentrations, no matter what sampling stategy was employed. The small increases in chlorophyll a levels observed at the inner two stations at both locations during specific sampling periods could have resulted as a consequence of three possible mechanisms, including:

1) Increases in growth rates of nitrogen-limited phytoplankton entering waters containing high ammonium levels as a result of fish farming activity;

2) Retention of phytoplankton in the immediate vicinity of the farms through the formation of eddies and gyres created as currents flow through the pens and rafts of the culture facilities;

3) Disengagement of benthic algae growing on the nets and floats of the fish farm.

The elevated chlorophyll a concentration observed at Station 0 at Yellow Island during the late June sampling period was at least partially caused by mechanism 3. This is supported by the large proportion of benthic diatom species observed during this time period. The percentage of benthic diatoms at Quartz Bay was also highest during the first sampling period; however, this group made up a much smaller proportion of the total biomass. Thus disengagement of fouling algae at Quartz Bay did not contribute significantly to the elevated chlorophyll a levels observed at Station 0 during early July and mid-August. If the total ammonia levels measured at Stations 2 and 3 during periods when minor chlorophyll a enrichment was observed are representative of background levels in Quartz Bay (*i.e.* 1-3 *u*M $NH_4$-N) during this time, it seems unlikely that higher levels of ammonium in the near vicinity of the farm would result in elevated chlorophyll a concentrations. By elimination of the first and third hypotheses, the minor increases in phytoplankton biomass in the vicinty of the pens at Quartz Bay probably resulted from the formation of retentive gyres produced by currents flowing past the culture facility.

The length of time for a change in phytoplankton community structure to occur due to changes in nutrient levels is many times greater than the time scale of algal metabolic processes. Thus it is not surprising that no spatial patterns of species dominance emerge between the inner and outer stations (Fig. 11) due to any existing gradients in total ammonia levels. The species composition data do shed some light on the importance of seasonal changes occurring in the water column. At both Yellow Island and Quartz Bay, the percentage of flagellates and ciliates was highest during the first sampling periods in late June/early July. The community shift at Yellow Island from 85 to almost 100% diatom dominance in the August and September sampling periods may reflect increasing ambient nutrient levels favoring larger cells with higher nitrogen uptake rates (Parsons *et al.*, 1977). At Quartz Bay, the increase in diatom abundance was more dramatic, with abundances changing from 9% in early July to roughly 95% in the last two sampling periods. The higher productivity indices seen in the latter two sampling periods at Quartz Bay (Fig. 10B) may, in part, have resulted from this marked composition shift.

Dissolved organic carbon absorbances (i.e. 280 nm absorbances) were quite low at both locations. The absorbance values observed over the course of the summer at both locations are approximately equivalent to 0.5 mg $C \cdot l^{-1}$

(T.R. Parsons, Pers. comm.). DOC absorbance at the outer two stations of Yellow Island (Table 8) was significantly higher ($p<.05$) than at the inner stations. Because the absorbance levels were so low it must be realized that the spatial trend seen at Yellow Island is occurring over a very small range of values. It is difficult, therefore, to determine if this pattern is a reflection of a significant negative influence of the farm on DOC levels in the immediate vicinity, or simply a measure of small scale patchiness.

Bacterial numbers in the surface waters at Yellow Island and Quartz Bay did not vary with respect to distance from the farms. If the low absorbance values (280 nm) are representative of the amount of available protein, it is not surprising that no elevation in bacterial numbers was observed in the vicinity of the farms. Bacterial concentrations did not decline in any of the pens in which antibiotic treatments were being administered suggesting an adequate dilution of any antibiotic leaching from the feed during settling. The larger numbers of bacteria at Quartz Bay compared to Yellow Island are probably the result of higher water temperatures as suggested by the positive correlation between bacteria and surface temperature ($p<.01$) seen at Quartz Bay (Table 10). Bacterial levels at both locations appear to be controlled by seasonal changes in the water column and not by fish farm by-products.

The main objective of this study was to determine if salmon farms in the Discovery Passage area are increasing levels of phyto- and bacterioplankton due to ammonium and/or carbon loading. This investigation has shown that salmon farms in this area can elevate levels of phytoplankton biomass in the immediate vicinity of the farms but only to a very limited extent. In terms of the production measures investigated, any influence of the farms appears to be lost within a very short distance (*ca.* 10 m) from the pens. If the sites examined in this study are at all representative of the majority of culture facilities in this region, a decline in water quality through elevations in phytoplankton levels resulting from fish farming activities seems highly unlikely in the Discovery Passage area. The magnitude and variability of any increases in total ammonia levels have been shown to be closely related to the flushing characteristics of the location as well as seasonal changes in the stability and nutrient availability of the surrounding water. Given the limited size of the sampling program employed, these conclusions should be viewed as preliminary in nature. The variability in any enriching effects of culture operations on marine water quality should nonetheless be an important consideration in the design and implementation of any upcoming monitoring requirements.

## VI. REFERENCES

Anonymous. 1988. Canadian Tide and Current Tables. Vol. 6. Barkley Sound and Discovery Passage to Dixon Entrance. Department of Fisheries and Oceans, information and publications branch.

Arakawa, K.Y. 1973. Aspects of eutrophication in Hiroshima Bay viewed from transition of cultured oyster production and succession of marine biotic communities. *Nihon Kaiyo Gakkai-Shi* 11: 43-48.

Arizono, M. 1979. Disease control in mariculture, with special reference to yellowtail culture. In G. Yamomoto (ed.), Proc. Seventh Japan-Soviet Joint Symposium on Aquaculture, September, 1978, Tokyo and Tsuruga, Japan., Tokai University. pp. 79-88.

Austin, B. 1975. Antibiotic pollution from fish farms: effects on aquatic microflora. *Microbiol. Sci.* 2: 113-117.

Azam, F., T. Fenchel, J.G. Field, J.S. Gray, L.A. Meyer-Reil and F. Thingstad. 1983. The ecological role of water-column microbes in the sea. *Mar. Ecol. Prog. Ser.* 10: 257-263.

Bergheim, A. and A.R. Selmer-Olsen. 1978. River pollution from a large trout farm in Norway. *Aquaculture* 14: 267-270.

Black, E.A. and B.L. Carswell. 1986. The impact of salmon farming on marine water quality. Fisheries Development Paper No. 11 (draft).

Buckley, J.R. and S. Pond. 1976. Wind and the surface circulation of a fjord. *J. Fish. Res. Bd. Canada.* 33: 2265-2271.

Cochlan, W.P., P.J. Harrison, P.A. Thompson and T.R. Parsons. 1986. Preliminary observations of the summer production of three British Columbia coastal inlets. *Sarsia* 71: 161-168.

Eloranta, P. and A. Palomaki. 1986. Phytoplankton in Lake Konnevesi with special reference to eutrophication of the lake by fish farming. *Aqua Fenn.* 16: 37-45.

Ennell, M. and J. Lof. 1983. Environmental impact of aquaculture-sedimentation and nutrient loading from fish cage culture farming. *Vatten* 39: 364-375.

Ervik, A., P. Johannessen and J. Aure. 1985. Environmental effects of marine Norwegian Fish Farms: International Council for the Exploration of the Sea C.M. 1985/F:37. 13 p.

Fribourgh, J.H., F.P. Meyer and J.A. Robinson. 1969. Oxytetracycline leaching from medicated fish feed. U.S. Bur. Sport Fish Tech. Pap. 40. 7 pp.

Gowen, R.J., and N.B. Bradbury. 1987. The ecological impact of salmonid farming in coastal waters: A review. *Oceanogr. Mar. Biol. Ann. Rev.* 3: 563-575.

Hall, P. and O. Holby. 1986. Environmental impact of marine fish cage culture. International Council for the Exploration of the Sea C.M. 1986/F:46. 19 p.

Harris, G.P. 1988. Phytoplankton Ecology: Structure, Function and Fluctuation. Chapman and Hall, New York. 384 p.

Inoue, H. 1972. On water exchange in a net cage stocked with the fish, Hamachi. *Bull. Japanese Soc. Sci. Fish.* 38: 167-176.

Jacobsen, P. and L. Berglind. 1988. Persistence of oxytetracycline in sediments from fish farms. Aquaculture 70: 365-370.

Kadowaki, S., T. Kasedo and T. Nakazano. 1978. Continuous records of DO contents by cruising in the coastal culture fish farms. I. Relation between DO content and fish density in cages. *Mem. Fac. Fish.*, Kagoshima University 27: 273-280.

Kaspar, H.F., G.H. Hall and A.J. Holland. 1988. Effects of sea cage salmon farming on sediment nitrification and dissimilatory nitrate reductions. *Aquaculture* 70: 333-344.

Krom, M.D. and E.R. Sholkovitz. 1977. Nature and reactions of dissolved organic matter in the interstitial waters of marine sediments. *Geochim. Cosmochim. Acta.* 41: 1565-1573.

Larsson, U. and A. Hagstrom. 1982. Fractionated phytoplankton primary production, exudate release and bacterial production in a Baltic eutrophication gradient. *Mar. Biol.* 67:57-70.

Liao, P.B. and R.D. Mayo. 1974. Intensified fish culture combining water reconditioning with pollution abatement. *Aquaculture* 3: 61-85.

Lorenzen, C.J. 1963. Diurnal variation in photosynthetic activity of natural phytoplankton populations. *Limno. Oceanogr.* 8: 56-62.

Marvin, K.T. and R.R. Proctor. 1965. Stabilizing the ammonia-nitrogen content of estuarine and coastal waters by freezing. *Limno. and Oceanogr.* 10: 288-289.

Newell, B.S. 1967. The determination of ammonium in sea water. *J. Mar. Biol. Ass. U.K.* 47: 271-280.

Nishimura, A. 1982. Effects of organic matter produced in fish farms on the growth of redtide algae. *Bull Plankton. Soc. Japan.* 29: 1-7.

Paasche. E. 1988. Pelagic primary production in nearshore waters. In: T.H. Blackburn and J. Sorensen (eds.) Nitrogen Cycling in Coastal Marine Environments. John Wiley & Sons, New York. pp. 33-57.

Parsons, T.R., B.E. Rokeby, C.M. Lalli, and C.D. Levings 1989. Experiments on the effect of salmon farm wastes on plankton ecology (draft).

Parsons, T.R., D.G. Webb, H. Dovey, R. Haigh, M. Lawrence and G.E. Hopky. 1988. Production studies in the Mackenzie River-Beaufort Sea Estuary. *Polar Biol.*8: 235-239.

Parsons, T.R., Y. Maita and C.M. Lalli. 1984. A manual of chemical and biological methods for seawater analysis. Pergamon Press, Oxford, 173 p.

Parsons, T.R., M. Takahashi and B. Hargrave. 1977. Biological oceanographic processes. 2nd edition. Pergamon Press, Oxford, 332 p.

Pease, B.C. 1977. The effect of organic enrichment from a salmon mariculture facility on the water quality and benthic community of Henderson Inlet, Washington. Ph.D. thesis, University of Washington, Seattle, Washington.

Pond, S. and G.L. Pickard. 1983. Introductory dynamical oceanography. 2nd edition. Pergamon Press, Oxford, 329 p.

Rosenthal, H., D. Weston, R. Gowen and E. Black. 1988. Report of the *ad hoc* study group on "Environmental Impact of Mariculture". International Council for the Exploration of the Sea. Cooperative Research Report No. 154. 83 p.

Ruokolahti, C. 1988. Effects of fish farming on growth and chlorophyll a content of *Cladophera*. *Mar. Poll. Bull.* 19: 166-169.

Seki, H., A. Otsuki, Y. Hara, K.V. Stephens, C.D. Levings and C.D. McAllister. 1987. Dynamics of organic materials in the Campbell River estuary at the time of the spring bloom of phytoplankton. *Arch. Hydrobiol.* 111: 209-216.

Seki, H., A. Otsuki, S. Daigobo, C.D. Levings and C.D. McAllister. 1984. Microbial contribution to the mesotrophic ecosystem of the Campbell River estuary during summer. *Arch. Hydrobiol.* 102: 215-228.

Slawyk, G. and J.J. MacIsaac. 1972. Comparison of two automated ammonium methods in a region of coastal upwelling. *Deep Sea Res.* 19: 521-524.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry. 2nd edition. Freeman Press, New York, 859 p.

Storch, T.A. and G.W. Saunders. 1978. Phytoplankton extracellular release and its relation to the seasonal cycle in a eutrophic lake. *Limnol. Oceanogr.* 23: 112-119.

Thompson, R.E. 1981. Oceanography of the British Columbia coast. *Can. Spec. Publ. Fish. Aquat. Sci.* 56. 291 p.

Tucker, C.S., S.W. Lloyd and R.L. Busch. 1984. Relationships between phytoplankton periodicity and the concentration of total and unionized ammonia in channel catfish ponds. *Hydrobiologia* III: 75-79.

Turpin, D.H. 1983. Ammonium induced photosynthetic suppression in ammonium limited *Dunaliella tertiolecta* (Chlorophyta). *J. Phycol.* 19: 70-76.

Weston, D.P. 1986. The environmental effects of floating mariculture in Puget Sound. Wash. Dept. Fish. and Wash. Dept. Ecol. 148 p.

Wood, E.D., F.A.J. Armstrong and F.A. Richards. 1967. Determination of nitrate in seawater by cadmium-copper reduction to nitrite. *J. Mar. Biol. Ass. U.K.* 47: 23-31.

# VII. APPENDICES.

## APPENDIX I

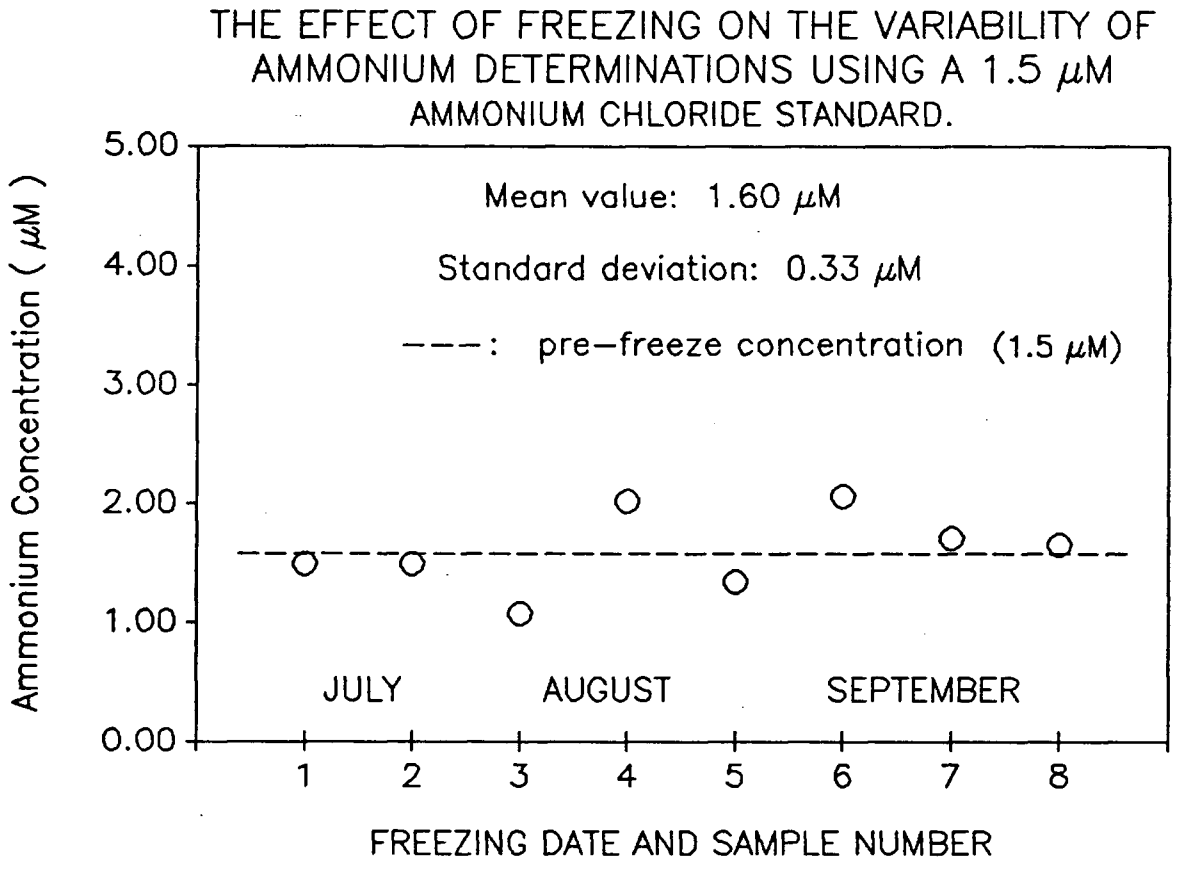
### THE EFFECT OF FREEZING ON THE VARIABILITY OF TOTAL AMMONIA DETERMINATIONS.

The magnitude of the effects of freezing on ammonium determinations is dependent on the concentration of total ammonia in the sample. At low concentrations typical of oceanic conditions (0.005-0.05 *u*M), freezing has been shown to cause substantial increases in the variability of determinations beyond that expected from measurement errors (Newell, 1967). At higher ammonium levels typical of coastal and estuarine environments (0.1-1.0 *u*M), freezing does not appear to cause significant changes in the variability of determinations (Marvin and Proctor, 1965). The high total ammonia concentrations measured in this study warrant the use of freezing as an adequate preservation technique.

To evaluate the effects of freezing on ammonium determinations made in this study, one 30 ml sample containing 1.5 *u*M $NH_4Cl$ prepared in 3% NaCl was frozen along with every daily set of nutrient samples. The results shown in Figure 16 suggest that freezing led to a 7% increase in total ammonia concentrations. The variability in the standard frozen determinations (coefficient of variation = 21%) also appears to be elevated beyond that expected from measurement error (coefficient of variation = 1.5%). The magnitude of the variability

attributed to freezing is not, however, great enough to result in the very large differences in total ammonia levels seen between the inner and outer station groupings during some of the sampling periods. The ammonium data therefore reflects real differences in concentrations at various distances from the pens, although the absolute ammonium values may be somewhat elevated due to the effects of freezing.

**Figure 16.** **The effects of freezing on ammonium determinations. The dashed line represents the concentration of the standard solution ($NH_4Cl$) before freezing.**



# APPENDIX II

## PHYTOPLANKTON SPECIES OBSERVED AT YELLOW ISLAND AND QUARTZ BAY.

Table 12 on the following pages contains all of the species observed in this study. Their presence at Yellow Island or Quartz Bay are denoted by the symbols Y and Q, respectively.

Class:
- Baci = Bacillariophyceae
- Cili = Ciliophora (Phylum)
- Cryp = Cryptomonads
- Dino = Dinophyceae
- Eugl = Euglenophyceae
- Raph = Raphidophyceae
- Sili = Silicophyceae

P/H = photosynthetic or heterotrophic.

Table 12. Organisms observed at Yellow Island and Quartz Bay, summer '88.

| ORGANISM OBSERVED | LOCATION | CLASS | P/H |
|---|---|---|---|
| Actinoptychus undulatus | Q, Y | Baci | P |
| Alexandrium ostenfeldii | Q, Y | Dino | P |
| Amphidinium sphenoides | Q, Y | Dino | H |
| Asterionella glacialis | Q, Y | Baci | P |
| Ceratium fusus | Q, Y | Dino | P |
| Ceratium longipes | Y | Dino | P |
| Cerataulina pelagica | Q, Y | Baci | P |
| Chaetoceros affine | Q, Y | Baci | P |
| Chaetoceros convolutum | Q, Y | Baci | P |
| Chaetoceros compressum | Q, Y | Baci | P |
| Chaetoceros constrictum | Q, Y | Baci | P |
| Chaetoceros danicum | Q, Y | Baci | P |
| Chaetoceros debile | Q, Y | Baci | P |
| Chaetoceros decipiens | Q, Y | Baci | P |
| Chaetoceros spp. | Q, Y | Baci | P |
| Chaetoceros laciniosum | Q, Y | Baci | P |
| Chaetoceros pseudocrinitum | Y | Baci | P |
| Chaetoceros radicans | Q, Y | Baci | P |
| Chaetoceros sociale | Q, Y | Baci | P |
| Corethron criophilum | Q | Baci | P |
| Coscinodiscus spp. | Q, Y | Baci | P |
| Coscinodiscus granii | Y | Baci | P |
| Coscinodiscus radiatus | Q, Y | Baci | P |
| Coscinodiscus thorii | Q | Baci | P |
| Cryptomonads | Q, Y | Cryp | P |
| Cylindrotheca closterium | Q, Y | Baci | P |
| Detonula pumila | Q, Y | Baci | P |
| Dictyocha speculum | Q, Y | Sili | P |
| Dinophysis infundibulum | Q | Dino | P |
| Dinophysis lachmannii | Y | Dino | P |
| Dinophysis norvegica | Q, Y | Dino | P |
| Dinophysis ovum | Q, Y | Dino | P |
| Dinophysis rotundata | Q, Y | Dino | H |
| Diplopsaloids | Q, Y | Dino | H |
| Ditylum brightwellii | Q, Y | Baci | P |
| Ebria tripartita | Y | Sili | H |
| Eucampia zoodiacus | Q, Y | Baci | P |
| Eutreptiella sp. | Q, Y | Eugl | P |
| Fragilaria spp. | Q, Y | Baci | P |
| Gonyaulax spinifera | Q, Y | Dino | P |
| Grammatophora marina | Y | Baci | P |
| Gymnodinium sanguineum | Q, Y | Dino | P |
| Gyrodinium glaucum | Q, Y | Dino | H |
| Gyrodinium spirale | Q, Y | Dino | H |
| Heterosigma akashiwo | Q, Y | Raph | P |
| Heterocapsa triquetra | Q, Y | Dino | P |
| Leptocylindrus danicus | Q, Y | Baci | P |

Table 12 con't.

| ORGANISM OBSERVED | LOCATION | CLASS | P/H |
|---|---|---|---|
| Leptocylindrus minimus | Q, Y | Baci | P |
| Licmophora spp. | Q, Y | Baci | P |
| Melosira moniliformis | Q, Y | Baci | P |
| Mesodinium rubrum | Q, Y | Cili | P |
| Navicula spp. | Q, Y | Baci | P |
| Nitzschia delicatissima | Q, Y | Baci | P |
| Nitzschia pungens | Q, Y | Baci | P |
| Nitzschia seriata | Y | Baci | P |
| Nitzschia spp. | Q, Y | Baci | P |
| Noctiluca scintillans | Q, Y | Dino | H |
| Odontella aurita | Q, Y | Baci | P |
| Odontella longicruris | Y | Baci | P |
| Oxyphysis oxytoxoides | Y | Dino | P |
| Paralia sulcata | Q, Y | Baci | P |
| Pleurosigma/Gyrosigma | Q, Y | Baci | P |
| Polykrikos kofoidii | Q, Y | Dino | H |
| Protoperidinium acutum | Y | Dino | H |
| Protoperidinium bipes | Y | Dino | H |
| Protogonyaulax catenella | Q, Y | Dino | H |
| Protoperidinium conicum | Q, Y | Dino | H |
| Protoperidinium denticulatum | Q | Dino | H |
| Protoperidinium depressum | Q, Y | Dino | H |
| Protoperidinium excentricum | Q, Y | Dino | H |
| Protoperidinium furcatum | Q, Y | Dino | H |
| Prorocentrum gracile | Q, Y | Dino | H |
| Protoperidinium granii | Y | Dino | H |
| Protoperidinium pallidum | Q, Y | Dino | H |
| Protoperidinium pellucidum | Q, Y | Dino | H |
| Protoperidinium pentagonum | Q, Y | Dino | H |
| Protoperidinium subcurvipes | Q | Dino | H |
| Protoperidinium subinerme | Q, Y | Dino | H |
| Protoperidinium spp. | Q, Y | Dino | H |
| Rhizosolenia fragilissima | Q, Y | Baci | P |
| Rhizosolenia setigera | Q, Y | Baci | P |
| Scrippsiella trochoidea | Q, Y | Dino | P |
| Skeletonema costatum | Q, Y | Baci | P |
| Stephanopyxis turris | Q | Baci | P |
| Thalassiosira aestivalis | Q, Y | Baci | P |
| Thalassiosira angstii | Q, Y | Baci | P |
| Thalassiosira anguste-lineata | Q, Y | Baci | P |
| Thalassiosira eccentricus | Q, Y | Baci | P |
| Thalassiothrix frauenfeldii | Q, Y | Baci | P |
| Thalassionema nitzschioides | Q, Y | Baci | P |
| Thalassiosira rotula | Q, Y | Baci | P |
| Tropidoneis lepidoptera | Y | Baci | P |