A comparison of some environmental costs associated with netpen culture of fish with some other forms of food production

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ABSTRACT

There are environmental costs associated with every form of food production and none of these appear sustainable at the present time. The goal of a rational society should be to achieve sustainability of all food production and use of natural resources. That means understanding and managing the environmental costs associated with all forms of agriculture, aquaculture and the harvesting of wild stocks of fish. Those costs should then be prioritized and society should focus its energies on efficiently solving the most demanding problems. In a global sense, those most demanding problems likely involve topsoil losses and the availability of fresh water. It has been obvious for several decades that the oceans' food resources are being over-exploited and few jurisdictions have been successful in managing the harvest of fish and shellfish. Aquaculture holds a promise to supplement the ocean's bounty. Small scale aquaculture is an ancient practice, but intensive aquaculture is relatively new and because of its scale, it can potentially carry significant environmental costs which must be managed to insure that they do not become widespread or irreversible. Aquaculture's emergence as a major source of seafood has created social and economic tensions within some societies that are played out as environmental issues using terms such as sustainability. This paper describes the near-field environmental response to organic enrichment associated with salmon aquaculture in the Northeast Pacific and a discussion of the sea lice issue in British Columbia. The major environmental cost identified to date is benthic enrichment. Significant effects appear to be restricted to a few hectares within 60 m of netpens. Minor effects have been seen to distances of 200 m. Chemical remediation of sediments at reasonably well sited farms takes six months to a year. In the worst case studied, chemical remediation was nearly, but not totally, complete after 5 years in fallow. This site was predicted to be chemically remediated seven years after production was halted. Biological remediation, as defined herein, appears to occur within a year following completion of chemical remediation. This analysis suggests that the empirically measured reductions in the biomass of benthic invertebrates results in the loss of approximately 300 kg of wild fish during production of 2,500,000 kg of Atlantic salmon. The yield of edible flesh from Atlantic salmon is 50% of live weight and it is 42% for beef cattle. In contrast to the small (1.6 Ha average) and ephemeral (44 month long) effects created by salmon farming, the growing of an equivalent amount of beef is shown to require 3.504 Ha of high quality pasture for 30 months plus as long as several hundred years for remediation. Sea lice have been observed on pink salmon fry in the Broughton Archipelago and it has been hypothesized that these stocks of fish are in jeopardy of extinction because of these infections. Furthermore, it has been asserted that the lice must have originated on farmed salmon because *L. salmonis* is specific to salmonids and Atlantic salmon were the only salmonids present in the estuary during spring. It is now known that three-spine sticklebacks, which are very abundant and ubiquitous in the archipelago, host both *Lepeophtheirus salmonis* and *Caligus clemensi*. In addition, it is known that resident salmonids are present in the archipelago year around. Models implicating salmon farms as the source of infective stages of lice have not included the life history of the louse, nor have they adequately addressed survival and dispersal of the pest. Building on the dispersal and life history issues raised in the

Broughton Archipelago, computer modeling that includes life history factors important to estimating the survival and dispersal of sea lice larvae has recently been field tested in Scotland. Those authors predicted that infective copepodids would be found 7 to 12 km from the local salmon farm. Field collection of nauplii and copepodids confirmed these predictions. Thus the observation of higher intensities of sea lice immediately adjacent to salmon farms is most likely either an artifact of the collection methods or an indication that the source of the infective lice was further inland in the estuary. Studies over the last four years have revealed that:

- Sea lice infections increase in years when surface seawater temperatures are slightly higher and that the distribution of competent sea lice larvae is dependent on salinity. Sea lice were absent or had a consistently low abundance in those zones where surface salinity was lowest.

- Surface salinity in the archipelago is significantly reduced by estuarine circulation to <30 practical salinity units (PSU) in late spring and summer. The pre-infective larval stages of *Lepeophtheirus salmonis* have been reported to not develop to an infective copepodid stage at salinities <30 PSU.

- British Columbia's Sea Lice Action Plan requires monitoring of cultured fish and control of sea lice using Slice™ when lice levels reach prescribed benchmarks. There is now a four year record of the abundance, prevalence and intensity of sea lice on cultured fish demonstrating that infections are lowest when salinity is low and that the Action Plan has reduced the prevalence and intensity of gravid female lice on cultured fish to very low levels.

- Pink salmon have varied significantly from year to year since at least 1954, decades before there was any marine aquaculture in British Columbia. These returns continue to fluctuate, but there is no evidence that returns to Atlantic salmon producing areas are significantly affected by the presence of the farms.

- Laboratory research indicates that pink salmon fry mount an effective immune response to sea lice infections and that healthy fry tolerate the low to moderate intensity infections observed in the archipelago.

The science regarding the importance of farmed salmon to infection of pink salmon fry remains contradictory. However, the accumulating record indicates that cultured Atlantic salmon are a controllable source of sea lice larvae in the Broughton Archipelago. Their relative contribution of lice larvae in comparison with natural sources has not been determined because the population of sea lice on wild hosts is unknown. It should be noted that these studies have been conducted in the presence of an aggressive government program designed to minimize the contribution salmon farms might make to the overall lice population in the archipelago. In British Columbia, the sea lice issue that was originally touted as a pending crisis has now become simply an interesting scientific question that is slowly being answered. The bottom line is that pink and chum salmon returns to the archipelago remain variable from year to year within the range observed prior to initiating salmon farming in the archipelago.

Achieving sustainability requires prioritizing the costs of all forms of food production and focusing our energy on solving the most important and tractable issues first. For instance, bicatch and lost fishing nets and pots waste a significant portion of the sea's bounty each year. From a sustainability point of view, these costs represent a far greater hazard to marine life than the lost production under a salmon farm.

Keywords: Sustainability; food production; salmon aquaculture; organic enrichment; terrestrial agriculture, sea lice, British Columbia.

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1.0. Introduction. As the expanding human population places additional stress on earth's resources, there is increasing interest in understanding and managing all of the costs of food production. Aquaculture is an ancient practice that holds a promise to supplement the ocean's supply of fish and shellfish in meeting the increasing demand for seafood. For many, this increased supply is seen as critical to proper management and conservation of the ocean's living resources. However, as with all human activity, the increasing intensity and scale of aquaculture has raised concerns that it may diminish natural productivity as it increases the human food supply. There are environmental costs associated with a loaf of bread. Figures1a and 1b describe soil erosion in Eastern Washington State where significant quantities of wheat are grown and the accumulation of this eroded topsoil behind one of several dams on the Columbia River. The annual soil loss from cropland in the United States is four tons/acre-year and depending on the global region, topsoil is being lost 16 to 300 times faster than it is being replenished and forty percent of earth's cropland is degraded (NRCS, 1999). During mediation of a major pesticide issue in Washington State, a farmer lamented that the pesticides he used to grow wheat were not tolerated by peas in their rotation. His solution was to flush the pesticides from his 6,000 acres of cropland with 15 cm of irrigation water which entered local salmon spawning rivers tributary to the Columbia. Both of these are examples of the environmental costs associated with a loaf of bread and discussions of the environmental costs of terrestrial agriculture are incomplete unless they include these and similar costs.

a) Local soil erosion

b) accumulations of eroded soil in behind a Columbia River dam

Figure 1. a) Local soil erosion in Eastern Washington. b) Accumulation of eroded topsoil behind one of the dams on the Columbia River in Washington State.

Aquaculture creates environmental costs as well. The first purpose of this paper is to assess the nearfield effects associated with organic enrichment from salmon aquaculture in the Northeast Pacific and to attempt to put them into perspective with some of the costs associated with producing an equivalent amount of beef. This information is provided in significant detail to inform the reader of the level of scientific rigor necessary to address these issues. Figure 2 describes a typical salmon production cycle lasting 32 to 34 months. The following definitions are used in this paper:

Nearfield effects are those that can be measured during point in time surveys. They include organic enrichment of sediments measured through sulfide, redox potential, nutrients (N and P) and organic matter measured as Total Volatile Solids (TVS) or Total Organic Carbon (TOC). Macrobenthic community surveys provide sensitive indications of environmental effects in homogeneous substrates. However, they are expensive and time consuming and they are problematic in heterogeneous environments. Near-field benthic effects have been observed to distances of 205 m downcurrent from Northeast Pacific salmon farms (Brooks, 2001). Water column effects can often be measured within netpens and to several meters downcurrent. However, they have not been detectable 30 m downcurrent at salmon farms in the Northeast Pacific (Brooks and Mahnken, 2003a). Nearfield effects have typically been monitored by producers in compliance with government mandated programs.

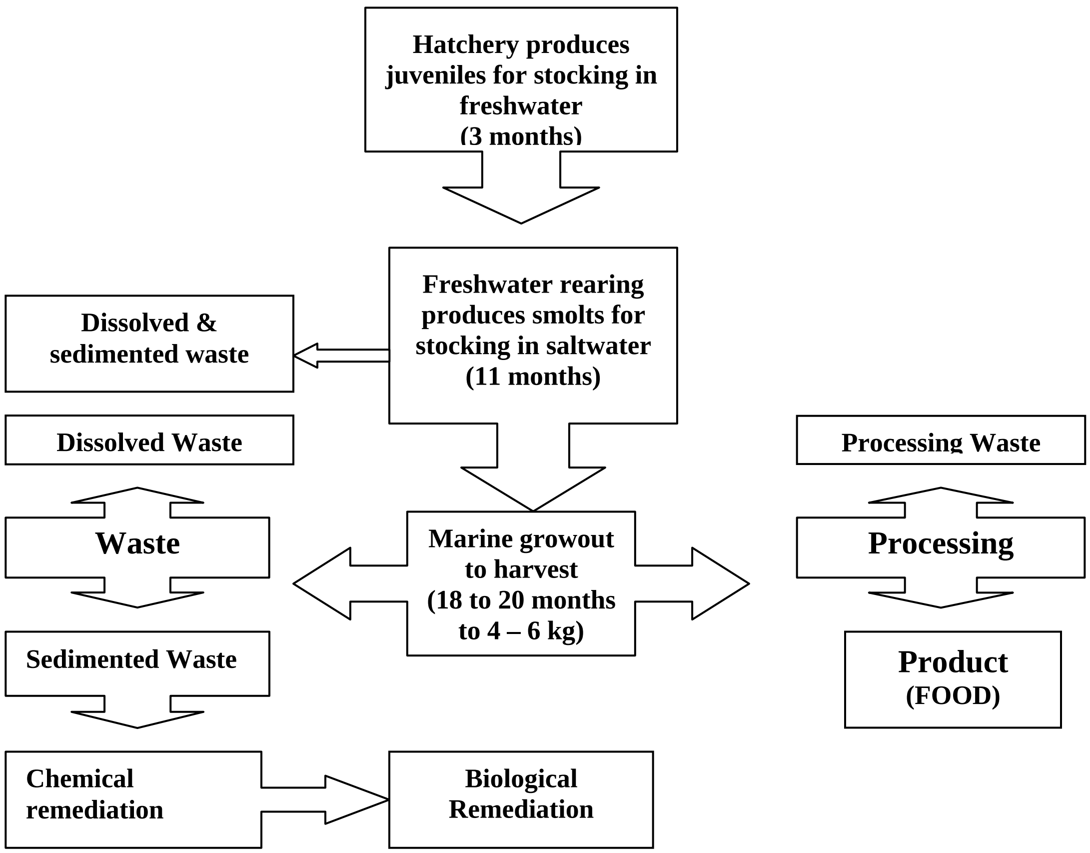

Figure 2. Phases in the production of Atlantic salmon and associated waste streams. A more inclusive list of potential hazards and environmental costs is provided in Appendix 1.

Far-field effects are those that cannot be measured by point in time surveys. These effects include ecosystem eutrophication resulting in increased primary production; reduced oxygen tension due to cumulative effects of all organic inputs, including salmon farms, and the biological oxygen demand associated with sedimented waste and the senescence of increased system-wide primary production. Assessment of far-field effects requires long-term and widespread monitoring to include collection of baseline data. Many coastal waters receive significant nutrient inputs from terrestrial activities including urbanization and agriculture. Therefore, determining cause and effect relationships between observed far-field effects and specific sources is difficult. Mass balance models are useful in this regard, but require inventories of (for instance) all nutrient inputs, which can be difficult and expensive in large and/or complex landscapes. There is increasing interest in waterbody specific computer models that track the dispersion of nutrients and other contaminants and their uptake by macroalgae and/or phytoplankton. Few waterbodies have been modeled in this regard, but the models allow government management by partitioning the total maximum daily load

(TMDL) among contributors. Monitoring and managing of waterbodies and the multitude of contributors of any stressor is usually undertaken by government. However, there are few examples of the application of this approach to aquaculture.

Environmental cost as used in this paper is defined as an imposed change that reduces the environment's natural productivity including the abundance and diversity of plants and animals within the affected area. Environmental costs are multidimensional in that they may create effects over some three dimensional space. They also have a temporal dimension. Ephemeral costs may last a few weeks to a few years. Longer term costs may reduce natural productivity for decades, and irreversible costs create changes that affect an environment's productivity for a century or longer. Lastly, environmental costs differ in the degree of their effects. The loss of natural productivity may be barely distinguishable or it can be dramatic resulting in near defaunation of an area. The "environmental cost" associated with an activity depends on all of these dimensions and as stated in the precautionary approach, the costs of greatest concern are those that are "significant, widespread and irreversible."

2.0. Categorizing the environmental costs associated with salmon aquaculture.

Cost and hazard analyses are site specific. Subtle environmental differences between and within regions require individual analyses. As will be seen, upwelling delivers large quantities of nutrients to nearshore areas in the Northeast Pacific with the result that primary production is generally light limited and is seldom nutrient limited. In this region, sedimented organic waste is the primary hazard observed during the marine growout phase of salmon production. Within the Northeast Pacific Region, there is tremendous variation in the extent and consequences of organic loading. Five to 10 percent of historic farms have created significant negative effects that have proven long lasting with chemical remediation taking as long as seven to ten years (Brooks *et al.*, 2004). At very well flushed sites with current speeds up to 125 cm/sec, the abundance and diversity of the macrobenthos has been significantly enhanced in response to salmon production (Brooks, 1995c). Varying degrees of adverse effects have been documented within 60 to 200 meters of netpens at perhaps 75 to 85 percent of Northeast Pacific sites. Sediments at several of these sites have been shown to chemically remediate in six months to a year (Brooks 1993, 1999, Brooks *et al.*, 2003c). Thus, while it is possible to discuss regional environmental costs in general, quantitative assessments of near-field effects must be conducted on a site specific basis.

Categorizing environmental costs. The environmental costs associated with any activity are, in large part, dependent on how the activity is managed and assessing environmental costs must be accomplished within the range of management options available. Definitions and typical management approaches for the following four types of hazards are provided below.

<u>Category</u> I hazards are common to many activities in coastal environments. These are costs associated with hazards that can be minimized or avoided through known strategies such as proper engineering, worker training, inspection of infrastructure, and etc. Examples of Category 1 hazards include collision of boats with aquaculture structures, which can be mitigated by proper lighting and other programs administered by government agencies such as the Coast Guard; avoidance of fuel spills; collection and disposal of trash, including feed bags; requirements for properly engineered anchoring systems; periodic inspection of infrastructure including containment nets; noise abatement; and etc. Category 1 hazards do require some level of risk assessment because the environmental exposure to them can vary significantly from site to site. Management of

Category 1 hazards is typically accomplished through imposition of *Conditions* on permits, use of *Best Management Practices, Codes of Conduct,* and government regulatory programs applicable to a broad range of coastal users.

Category II environmental hazards are inherent to the intensive cultivation of all plants and animals. They include organic enrichment from fed aquaculture (shrimp and piscivorous fish) and organic depletion associated with extractive aquaculture (bivalves, carp, etc). For fed aquaculture, it is the local area's *assimilative capacity* that is challenged and for extractive aquaculture it is the *carrying capacity* of the system that must be considered. These hazards can result in either positive or negative effects. In some cases enrichment may result in increased abundance and diversity of wildlife. As the degree of enrichment increases beyond the environment's assimilative capacity, negative responses associated with eutrophication including reduced sediment redox potential may occur. Similarly, extractive aquaculture may be critical to controlling eutrophication in some estuaries that are naturally or anthropogenically enriched. Chesapeake Bay in the United States is an excellent example of an estuary suffering from the lack of the extraction of phytoplankton by bivalves (Newell, 1988). However, the Bay of Marennes-Oleron is an example of an estuary in which overstocking of oysters resulted in exceeding the estuary's carrying capacity, causing reduced growth of the cultured species and likely reduced productivity of the entire food web (Raillard and Menesguen, 1994). In either case, Category II hazards are the inevitable result of the intensive cultivation of animals. While these hazards cannot be avoided in open culture, they can be managed to enhance environmental health in some cases and to control the temporal and spatial extent of adverse effects in others. Management typically begins with careful siting and restraints on allowable production levels at both the local and ecosystem levels. Computer models provide promise of assessing the environment's assimilative or carrying capacity on increasing spatial scales. However, these models have not yet achieved a level of sophistication providing reliable predictions of environmental (chemical or biological) responses. In many cases, the environmental effects associated with Category II hazards are managed through implementation of *Performance Standards.* Monitoring and enforcement is then required to insure compliance. Nearfield benthic effects, discussed herein, are an example of Category II costs.

Category III hazards are associated with potential, but not necessarily inevitable, release of contaminants. These hazards include sediment accumulations of trace metals originating in feed and antifouling compounds; therapeutants including antibiotics and pesticides; organic inputs associated with net cleaning, disposal of mortalities; and etc. Category 3 hazards are managed through proper siting and efforts to minimize or eliminate their effects. These hazards are frequently amenable to quantitative or semi-quantitative risk assessment and are managed by imposition of *Performance Standards*.

Category IV hazards are those for which the evidence is inconclusive or for which there the limited knowledge making quantitative or semi-quantitative assessments difficult or impossible. They involve disease transfer in both directions; ecological interactions (competition for habitat and food) associated with cultured shellfish and escaped finfish; and genetic interactions between cultured and wild species. Some of these interactions have been better studied than others (mussel genetics; transfer of disease from wild stocks to cultured stocks; disease transfer to both wild and cultured stocks of bivalves associated with poorly controlled movement of flat oysters (*Ostrea edulis*) resulting in *Bonamia* infections; or the spread of *Perkinsus marinus* and MSX in cultured and wild stocks of American oysters (*Crassostrea virginica*). Other Category IV hazards have not been well documented or remain controversial - such as the contribution of sea lice from Atlantic salmon cultured in the Northeast Pacific to wild stocks of pink salmon (*Oncorhynchus gorbuscha*) as discussed by Brooks (2005, 2006)) or the potential for Mediterranean mussels (*Mytilus galloprovincialis*) to displace the more common Baltic mussel (*Mytilus trossulus*) in the Northeastern Pacific (Brooks, 2005a). These types of hazards are difficult to assess quantitatively or semi-quantitatively and they are frequently studied only in an effort to develop management strategies when a need is observed. The potential for sea lice originating on farmed salmon to adversely affect populations of pink or chum salmon in British Columbia is an example of a Category IV hazard.

3.0. Environment costs associated with organic enrichment from salmon aquaculture. The locations of the 45 salmon farms included in the database upon which the following discussion is based are described in Figure 3. All of these sites have relatively cool water varying between ca. 6 or 7 ^{o}C in winter and 16 ^{o}C in summer. In 2003, British Columbia and Washington State produced approximately 65,000 mt of mostly Atlantic salmon (*Salmo salar*) and small amounts of coho (*Oncorhynchus kisutch*) and chinook (*Oncorhynchus tshawytscha*) salmon. Salmon farms in the Northeast Pacific are generally located in water depths of 18 to over 100 meters with average current speeds varying between 3 and >25 cm/sec. Maximum harmonically driven current speeds vary between 10 and 125 cm/sec.

Dissolved oxygen. Weston (1986) reviewed the effects of salmon culture on ambient concentrations of dissolved oxygen (DO) and concluded that salmon farms could decrease these levels by 0.3 ppm. Brooks (1991, 1993a, 1993b, 1993c, 1994a, 1994b, 1995a, 1995b, 1995c) observed decreases of as much as 2 ppm in water passing through a large, poorly flushed farm in Puget Sound. Statistically significant reductions in DO were not observed by Brooks (1994b, 1995b, 1995c) at farms in well-flushed passages. In no cases were DO levels within 6 m of the downstream farm perimeters' depressed below 6 mg/L, a minimum level for optimum culture of salmonids. Winsby *et al*. (1996) suggested that depressed oxygen levels were associated with the water column immediately overlying anaerobic sediments and that salmon farming had minimum potential to adversely oxygen concentrations in the water column. These results suggest that salmon farms do not currently impose a cost on Northeast Pacific environments associated with the consumption of oxygen. However, naturally depressed oxygen concentrations associated with upwelling have severely stressed cultured fish, leading, in a few cases, to mortality. This affects the overall environmental cost of salmon production because the dead salmon represent a wasteful sink of valuable resources associated with feed and other fixed costs that are not realized as human food.

Dissolved nutrient loading in the Northeast Pacific. Salmon and most other fish excrete 75 - 90% of their ammonia and ammonium waste across gill epithelia (Gormican 1989) or in concentrated urea (Persson 1988, and Gowen *et al.* 1991). Brett and Zala (1975) reported a constant urea excretion rate by sockeye salmon of 2.2 mg N/kg per hour. Nitrogen and phosphorus are also dissolved from waste feed and feces during and after their descent to sediments. All of these dissolved forms of nitrogen are readily available for uptake by phytoplankton. Silvert (1994a) suggested that 66 to 85% of phosphorus in feed is lost in a dissolved form to the environment at salmon farms. However, phosphorus is plentiful in Northeast Pacific marine environments (Figure 4) and seldom limits primary production (Brooks 2000a, 2006).

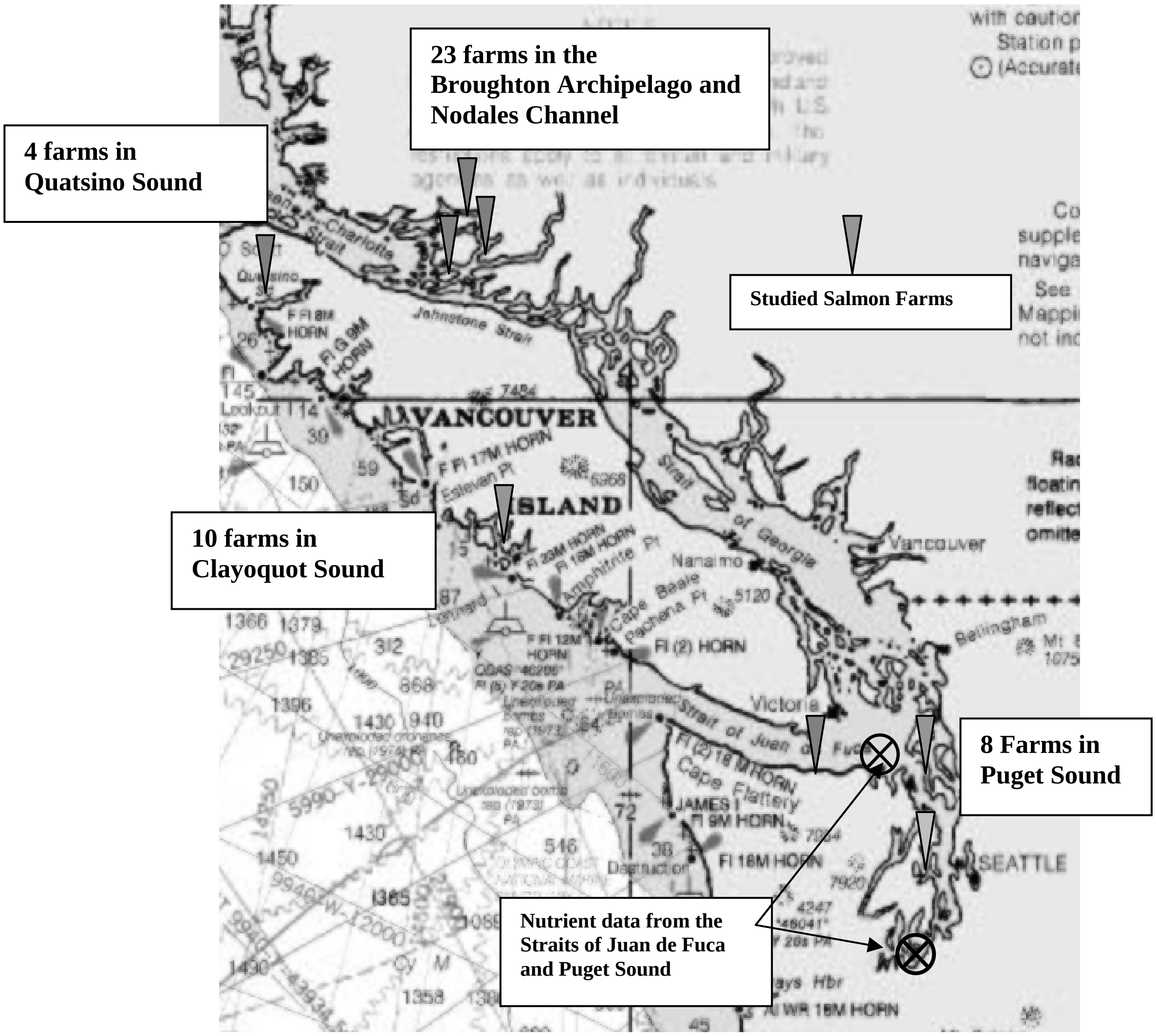

Figure 3. Site map describing the general location of 45 salmon farms monitored between 1991 and 2005 that have generated data forming the basis of the analysis presented herein.

Statistically significant increases in soluble nutrients at salmon farms have infrequently been observed in Puget Sound (Rensel 1989, and Brooks 1994a, 1994b, 1995a, and 1995b). Prior to 1995, Aquatic Lands Leases (ALLs) for salmon farms in Washington State required monitoring of NO_3, NO_2 and total ammonia (NH_3 + NH_4). Worst case concentrations observed between 1989 and 1995 are summarized in Table 2. Consistent with these results, monitoring by Pease (1977), Rensel (1988 and 1989), and Parametrix (1990) documented small increases in dissolved nitrogen within and on the perimeter of salmon farms. However, all of these authors agreed that the quantity of dissolved nitrogen added by even several farms would have no measurable effect on phytoplankton production. Gowen et al. (1988) studied a Scottish loch with restricted water exchange to the open sea and a large salmon farm. They concluded that the farm had no measurable effect on phytoplankton density. Similar results have been found in other salmon farming regions (Soto and Norambuena, 2004).

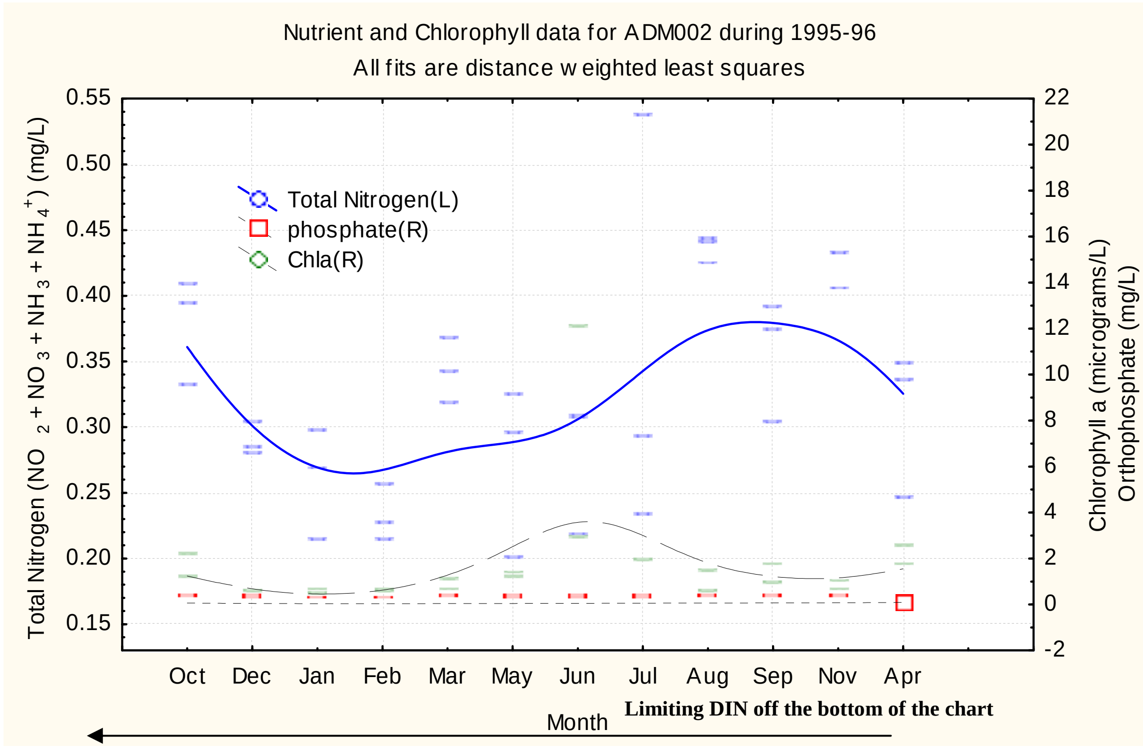

Figure 4. Dissolved inorganic nitrogen, orthophosphate and chlorophyll *a* recorded in the Straits of Juan de Fuca at Admiralty Head (WDOE, 1998, 2002).

Table 2. Water column dissolved inorganic nitrogen (µmoles/L) and unionized ammonia (in parentheses measured in mg/L) in the vicinity of salmon farms in Puget Sound, Washington (Brooks, 1993a, 1993b, 1993c, 1994a, 1994b, 1995a, 1995b, 1995c).

Farm	Dissolved Inorganic Nitrogen (µM)		
	100' upstream	20' downstream	100' downstream
A (1995)	9.58	14.87 (0.0002)	11.43
B (1994)	21.34	22.87 (0.0004)	23.04
C (1989)	22.51	25.83 (0.0003)	23.87
D (1994)	12.54	11.80 (0.0003)	12.15
E (1995)	5.47	5.16 (0.0002)	5.18
F (1995)	10.70	10.83 (0.00001)	11.85
G (1995)	6.06	6.21 (0.0002)	5.71
H (1994)	9.78	11.34 (0.0001)	10.80

In general, the variability between replicate samples taken at the 6 m downstream station was as great, or greater, than observed increases in nitrogen between upstream and downstream stations. No significant increases in nitrogen were observed at any of the 30 m downstream stations at any time. The greatest increase in reported DIN between upcurrent and downcurrent stations was 0.09 mg/L or 8% of the mean DIN values observed by Weston (1986) in Puget Sound and the highest observed level of toxic unionized ammonia (NH_3) reported by Rensel (1989) inside salmon netpens was 0.0004 mg/L, which is lower by a factor of 87.5 than the U.S. Environmental Protection Agency (USEPA) chronic exposure (4-day) concentration limit of 0.035 mg/L at pH = 8 and T = 15 ^{o}C when sensitive salmonid species are present.

Burd (1997) estimated that upwelling delivered approximately 2,000 mt of nitrogen to coastal British Columbia and Puget Sound environments each day. River inputs added 100 mt and sewage inputs were estimated at 70 mt. At that time British Columbia salmon farms were producing ca. 22,000 mt of salmon/year and it was estimated that they added another 6 mt of DIN/day. Scaling linearly to current production of 58,000 mt suggests that in 2005, salmon farms contributed ca. 15.8 mt DIN to coastal environments or 0.7% of the total 2,185.8 mt.

Other factors affecting primary production. In the Pacific Northwest, wind-driven vertical-mixing drives a significant proportion of the standing biomass of phytoplankton below the compensation depth where cell respiration equals photosynthesis and where phytoplankton populations no longer multiply. Where water freely circulates, flood tides replenish nutrients from offshore upwelled water. When coupled with the atmospheric and geographical factors that reduce light availability, the result is that primary productivity is generally light limited, not nutrient limited. This is especially true during winter months. In other words, there is insufficient light to use the nutrients already available in the water column. Adding nutrients to a light limited system does not increase plant growth. There are sheltered, poorly flushed, shallow embayments with long residence times (>10 to 20 days) where salinity and temperature induced stratification results in a stable water column allowing phytoplankton to remain above the compensation depth. When these conditions appear in the spring or summer, significant blooms can occur following several days or weeks of clear sunny weather. These blooms eventually wane because winds increase vertical mixing; cloud cover reduces the available light; or nutrients are depleted in the surface water. In this last situation, nutrient input from intensive aquaculture could further stimulate plant growth, exacerbating the problem. Shallow bays having significant freshwater input and minimal flushing, are not considered good sites for net-pen growout operations. However, they might be deemed appropriate as smolt introduction sites.

The last point to consider in this discussion is that nitrogenous compounds are released from fish farms into currents that generally average greater than 4 to 12 cm/sec and acoustic Doppler current meter studies at British Columbia salmon farms have revealed net transport (resting current) speeds of 1.0 to 5.0 cm/sec. At temperatures of 10 – 15 ^{o}C, it takes one to two days for an algal cell to divide, even if all of its photosynthetic needs are met (Brooks 2000). An algal bloom may result in cell densities increasing from a few thousand cells/ml to a million or more. That requires eight or nine cell generations, which takes a minimum of 8 - 16 days. In open bodies of water, moving with a net speed of 2 cm-sec^{-1}, a phytoplankton population would move 14 km from the location at which nutrients were added during creation of a bloom. Recall that the barely significant increases in nitrogen observed 6 m downstream from farms in Puget Sound were generally not detectable 30 m downstream. Within a single algal cell division (one to two days), the water passing through the farm would have traveled at least 1.7 km. It is difficult to conclude that nutrient additions from a farm, generally undetectable at 30 m downstream, would have any affect on primary production even if the water body was nutrient limited. Added nutrients can increase the growth of attached macroalgae in the immediate vicinity of salmon farms with increased abundance of nitrophilous green species such as *Ulva fenestrate* (Brooks, unpublished).

Supporting these theoretical arguments are studies conducted by Banse et al. (1990), Parsons *et al.* (1990), Pridmore and Rutherford (1992), Taylor (1993), Taylor *et al.* (1994), Taylor and Hatfield (1996) and Taylor and Horner (1994) who examined phytoplankton production and blooms of noxious phytoplankton in the Pacific Northwest and concluded that nitrogen levels and phytoplankton production at salmon farms were determined by ambient conditions and that aquaculture added little to the abundant nutrients supplied in upwelled water. These conditions are specific to the Northeast Pacific and the conclusions should not be extended to other regions without careful consideration. This issue was reviewed because it is an example of the importance of siting in minimizing the environmental costs associated with Category II hazards.

Benthic effects associated with solid waste. From an environmental point of view, it is sedimented waste that currently appears to carry the highest environmental costs in association with fed aquaculture. This is a Category II hazard that appears to create quantifiable and inevitable environmental costs in the near-field.

Waste feed. The amount of waste feed depends on feeding efficiency, which is principally influenced by feed composition, feeding methods, water currents at the site, and net-pen configuration. Beveridge *et al.* (1991) stated that up to 30% of feed was lost during the early years of salmon farming. Rosenthal *et al*. (1995) noted higher losses for wet feeds (up to 35%), than for dry feeds. Weston (1986) suggested that less than 5% of dry feed was lost at Puget Sound salmon farms. This is consistent with the research by Gowen and Bradbury (1987), who reported dry feed losses of 1 - 5%. Findlay and Watling (1994) reported maximum feed loss rates of between 5 - 11%, and that the average feed wastage was <5%. Dry and semi-moist feeds are now used exclusively in the Northeast Pacific and current feed loss rates are estimated at between 3 % and 5 % (J. Mann, EWOS Canada Ltd., personal communication). Modern monitoring systems incorporating feedback cones and underwater video or acoustical devices described by Mayer and McLean (1995) are now commonly used to monitor feeding behavior in efforts to minimize losses of uneaten feed from net-pens. Most of the current feed loss is associated with abrasion and breakage in automatic feeders, which can result in the disintegration of 4 - 5% of the pellets. Optimum feeding systems, with short delivery distances that are operated by compressed air valves, may reduce disintegration to <0.5% of the pellets (J. Mann, EWOS Canada Ltd., personal communication). The results of this review are reasonably consistent and indicate that at this time, 5% or less of the dry feed delivered to cultured salmon in net-pens is lost to the environment. These low rates are due to the combination of improved feedback technologies and the practice of quickly feeding the fish to satiation once or twice each day. Improvements in feed delivery systems to minimize pellet disintegration will probably reduce losses further. This assessment will assume feed losses are 5%. It should be noted that wasted feed is accounted for in the computation of the economic food conversion ratio (FCR), but not in the calculation of a biological FCR.

Fish feces. Weston (1986) estimated that 25 - 33% of the feed consumed by fish was ejected as feces. Modern diets are approximately 87 - 88% digestible (J. Mann, EWOS Canada Ltd., personal communication). The remaining ash consists primarily of calcium and inorganic phosphate, and represents 8.0 - 8.5% of the feed. This implies that approximately 12.5% of the dry weight of ingested feed will be ejected in feces. Subtracting the 87.7% that is digested and assimilated by the fish and 8.25% for ash, leaves about 4% of the feed that is ejected as labile organic material in the feces. If 5% of the feed is uneaten (Findlay and Watling 1994) and feces contribute organic matter equivalent to 4% of the feed weight, then approximately 8.8% of the labile organic compounds delivered in feed is discharged from the net-pen structure in particulate form, contributing to biological oxygen demand (BOD) in sediments.

Fish carcasses as organic wastes. Winsby *et al*. (1996) reviewed the mortality of fish at BC salmon farms in 1994. Their data suggested approximately 2,000 mt of salmon died at farms that year, or approximately 9% of the total production of 22,000 mt. They concluded that most of the salmon carcasses were removed to government approved compost disposal locations. No inappropriate disposal of salmon carcasses has been documented in the literature. Losses of farmed salmon are generally restricted to individual fish, which may have been attacked and killed by predators; died as a consequence of toxic algal blooms; or as a result of disease. Codes of Practice require physical removal of carcasses on a daily basis and therefore they do not contribute to BOD in the environment.

Quantification of solid organic waste from salmon aquaculture. Ackefors and Enell (1994) estimated the total organic output from salmon farms on the order of 2.5 mt wet weight/mt of fish produced. Gowen et al. (1991) cited three studies assessing the flux of carbon through salmon net-pens. In all three cases the harvested fish retained 21 - 23% of the carbon in feed and it was estimated that 75 - 80 % of the carbon was lost to the environment mostly in a dissolved form as CO_2. Merican and Phillips (1985) estimated that 35.6% of the carbon, 21.8% of the nitrogen, and 65.9% of the phosphorus were lost to the environment in solid form. Other estimates of the total suspended solids output from intensive net-cage culture of fish by Kadowaki et al. (1980), Warrer-Hansen (1982), Enell and Lof (1983), and Merican and Phillips (1985) range from 5 - 50 g suspended solids/m^2-day. All these publications are more than 15 years old and therefore these values do not reflect recent improvements in fish feed and feeding technologies.

Gowen and Bradbury (1987) estimated organic waste sedimentation rates of 27.4 g/m^2-day under Irish salmon farms, and an average of 8.2 g/m^2-day immediately adjacent to the perimeter of the net-pens. Gowen et al. (1988) measured average rates of 82.2 g dry weight/m^2-day on the perimeter of a net-pen in Washington, and Cross (1990) estimated an average overall sedimentation rate of 42.7 g TVS/m^2-day with a maximum of 94.5 g total volatile solids (TVS)/m^2-day at seven salmon farms in BC. More recent work by Findlay and Watling (1994) in Maine measured sedimentation rates on the perimeter of salmon farms at between 1.0 - 1.6 g carbon/m^2-day, and Hargrave (1994) summarized sedimentation rates from less than one to over 100 g carbon/m^2-day from salmon cage operations.

Brooks (2001) derived a theoretical estimate of contemporary TVS loading near fish farms. Given a feed with 11% moisture content and FCR of 1.2, the feed provided (1.2 kg x 89 % dry matter) or 1.07 kg dry feed/kg of fish produced. When coupled with the previously given estimate for the percent labile organic waste of 8.8% this equals 0.094 kg solid organic waste/kg of fish produced. A salmon farm producing 1,500 mt of salmon during a 16 to 20 month production cycle would therefore discharge 141 mt of particulate organic waste on a dry weight basis. Furthermore, assuming a fish density of 10 kg/m^3 in cages 15 m deep and a grow-out cycle of 18 months, the annual sediment load on average would be:

$$(10 \text{ kg fish/m3 x } 15 \text{ m deep x } 0.094 \text{ kg TVS/kg fish})/548 \text{ days} \quad = \quad 25.7 \text{ g TVS/m}^2\text{-day.}$$

The load would, in reality, be lower at the beginning of the grow-out cycle and increase towards maximum biomass. Brooks (2001) analyzed sediments collected in canisters deployed 5 m above the bottom at varying distances from seven farms in BC and at reference stations. The mean loading of volatile solids on the perimeter of these farms was 39.2 g TVS/m^2-day. The mean deposition of volatile material at the control stations was 6.3 g TVS/m^2-day and the contribution by the farm was approximately 32.9 g TVS/m^2-day. These studies were completed near peak salmon biomass and the observed values would therefore be greater than the theoretical average of 25.7 g TVS/m^2-day calculated above. Nonetheless, these observed and theoretical values are reasonably close.

Site specific models, such as DEPOMOD (Cromey *et al.*, 2002) are now used in British Columbia to predict the deposition of organic carbon associated with proposed salmon farms. Figure 5 is an example of the model's output. Several comparisons between DEPOMOD predictions and empirical evidence in the form of sediment physicochemical changes have been made by Brooks (unpublished). In general, these comparisons show remarkably similar patterns of responses when resuspension is turned off in the DEPOMOD program. The model only predicts deposition rates of organic carbon and it does not yet include modules predicting more meaningful physicochemical or biological responses.

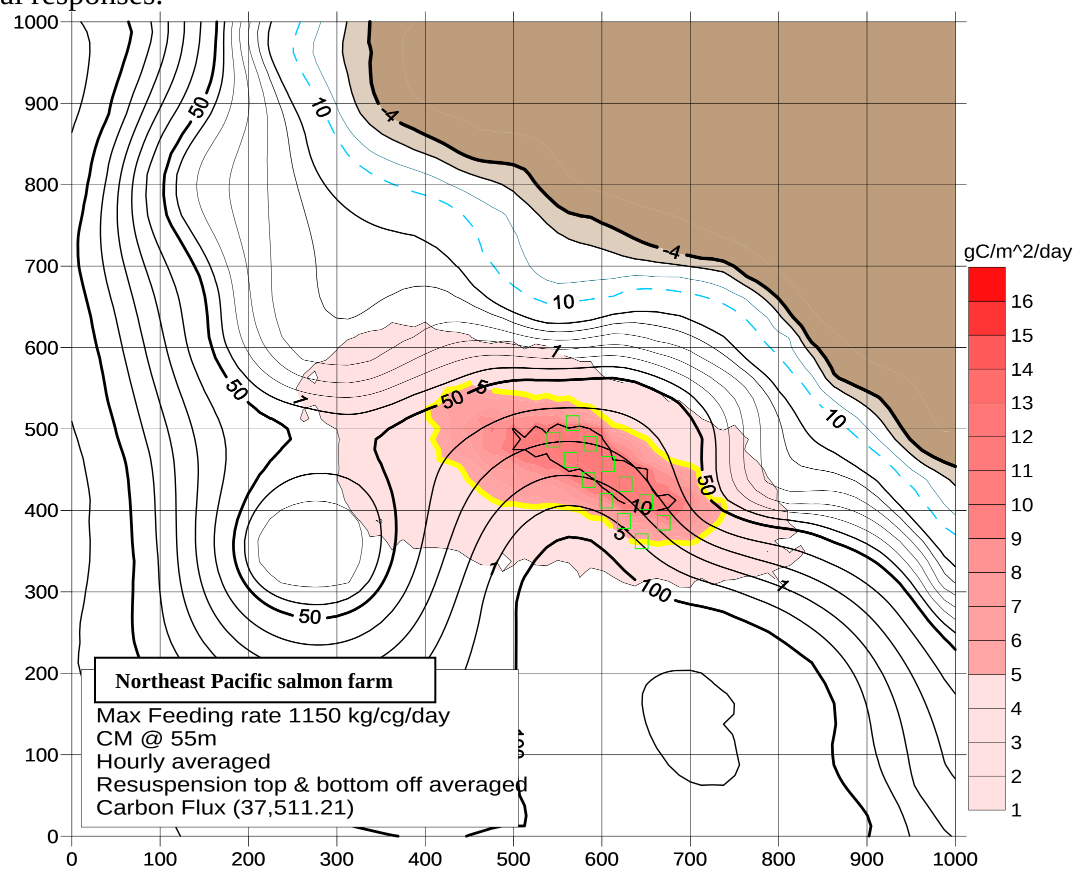

Figure 5. Output from the DEPOMOD program (Cromey *et al.*, 2002) for s salmon farm located over a rocky bottom in British Columbia.

Sediment physicochemical response to salmon farm inputs. Findlay and Watling (1994) developed a simple model for estimating aerobic carbon degradation rates (g C/m^2-d) based on the minimum two hour-average bottom current speed (cm/s). They estimated that at low bottom current speeds (<0.1 cm/s) a theoretical maximum aerobic degradation rate of ca 4.0 g C/m^2-d could be achieved. The predicted aerobic carbon degradation rate appears to asymptotically approach a value of ca. 22 g C/m^2-d at bottom current speeds greater than 10 to 12 cm/s. Time weighted 15 m deep current speeds at many B.C. salmon farms averaged 3.5 to 9 cm/s and the two hour minimum mean surface current speeds are generally < 3 cm/s (Brooks, unpublished). Even assuming that bottom current speeds equal near surface speeds, the model of Findlay and Watling (1994) predicts a maximum carbon assimilation rate of ca. 17 g C/m^2-d at 3.0 cm/s. The sedimentation rates reported by Brooks (2001) at seven British Columbia salmon farms generally exceeded this value and therefore it should be expected that the assimilative capacity of sediments in the vicinity of salmon farms is exceeded and that changes in sediment chemistry will occur while the excess carbon is being assimilated. Those effects are well document in a voluminous literature describing similar benthic responses from around the world. The following paragraphs describe sediment physicochemical responses to organic inputs recorded in this literature.

Organic content of sediments. Factors affecting the accumulation of waste include fish biomass and feeding rates; fish food and fecal material particle sizes and densities; netpen configuration; water depth; current speeds; and the degradation rate of sedimented carbon which depends primarily on the availability of oxygen and sulfate. The proportion of farm derived TVS observed in sediments integrates all of these factors. In addition to farm waste, there are numerous sources of natural TVS including terrigenous material, eelgrass and macroalgae, senescent plankton, etc. Many of these natural sources are refractory creating lower biological oxygen demand (BOD) than labile farm waste. As demonstrated by Brooks (2001), these differences in the nature of TVS confound the use of sediment carbon as an indicator of benthic effects.

There is a diverse literature describing sediment organic content adjacent to salmon farms in other parts of the world (Ye *et al.* 1991; Holmer and Kristensen 1992; Johnsen *et al.* 1993; Hargrave *et al.* 1995, 1997; Lu and Wu, 1998; Karakassis *et al.* 1999). These reports demonstrate consistent, but highly variable, increases in carbon under and immediately adjacent to salmon farms. This literature also suggests that waste deposits from fish farming are locally patchy with significant variability in replicates from the same sample station. Brooks (1999) described the spatial extent and temporal behavior of TVS in sediment adjacent to a British Columbia salmon farm that produced 1,200 mt of Atlantic salmon in 1996. Figure 6 describes the proportion TVS observed in sediments from just before peak biomass in August 1996 through a six-month fallow period, which ended in October 1997. Sediment TVS adjacent to the netpen perimeter declined rapidly from a peak of 35% at peak biomass to values indistinguishable from background by June 1997, three months following completion of harvest. Increased TVS extended to at least 75 meters. Samples were not collected beyond 75 meters because the sediment texture changed at that point from muddy sand to sandy gravel, which continually fouled the grab. Brooks (2000b) reported the results of evaluating 676 sediment samples collected at 34 British Columbia salmon farms between 1996 and 2000. The TVS data are summarized in Figure (7). Each of the large filled circles represents the TVS value equal to the upper 90th percentile observed at B.C. reference stations with percent fines (< 63 μm fraction) equal to 22.5% (lower right), 42.5% (center) or 62.5% (upper right). Exceedances of this 90th percentile TVS benchmark occurred at distances up to 80 meters in fine-grained sediments; to 60 meters at sites with ca. 42.5% fines and to 140 meters downcurrent from sites located in erosional environments with < 22.5% sediment fines. The biological implications of exceeding the upper 90th percentile TVS observed at a reference station sharing the same water depths and grain size distribution were not investigated in that study, but Brooks (2001) provides a detailed description of the macrobenthic response to a suite of physicochemical endpoints. Figure (7) strongly suggests that salmon farm effects extended beyond 100 m and Brooks (2001) found measurable, albeit small, effects at distances up to 205 m from farms near peak biomass.

Sediment oxidation reduction potential (Redox). Oxygen is delivered to sediments by diffusion from the overlying water column, and by mechanical infusion of overlying water into the sediments. This last transport mechanism is important in coarse-grained sediments with high porewater volume. Infusion is also enhanced by bioturbation. Mechanical infusion becomes less important as the sediment modal grain size decreases and likely has little affect on sediment redox potentials in fine-grained sediments containing >60% silts and clays. However, healthy infaunal

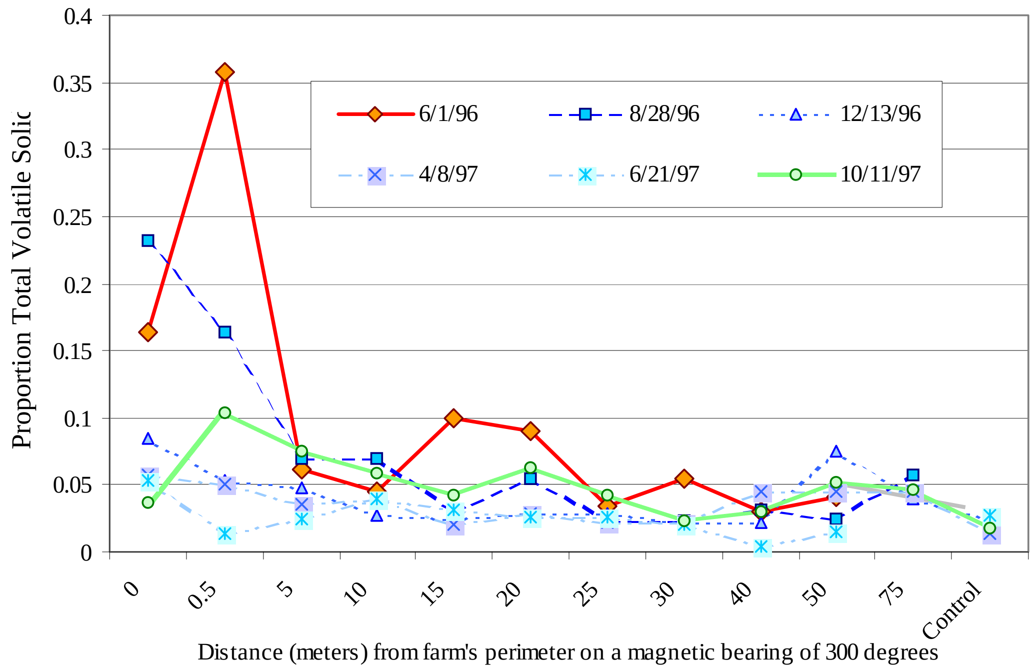

Figure 6. Total volatile solids observed in sediments under the Moonbeam salmon farm as a function of time (lines) and distance (meters) from the perimeter of the farm on the downcurrent transect (300 °M) and at a remote control station.

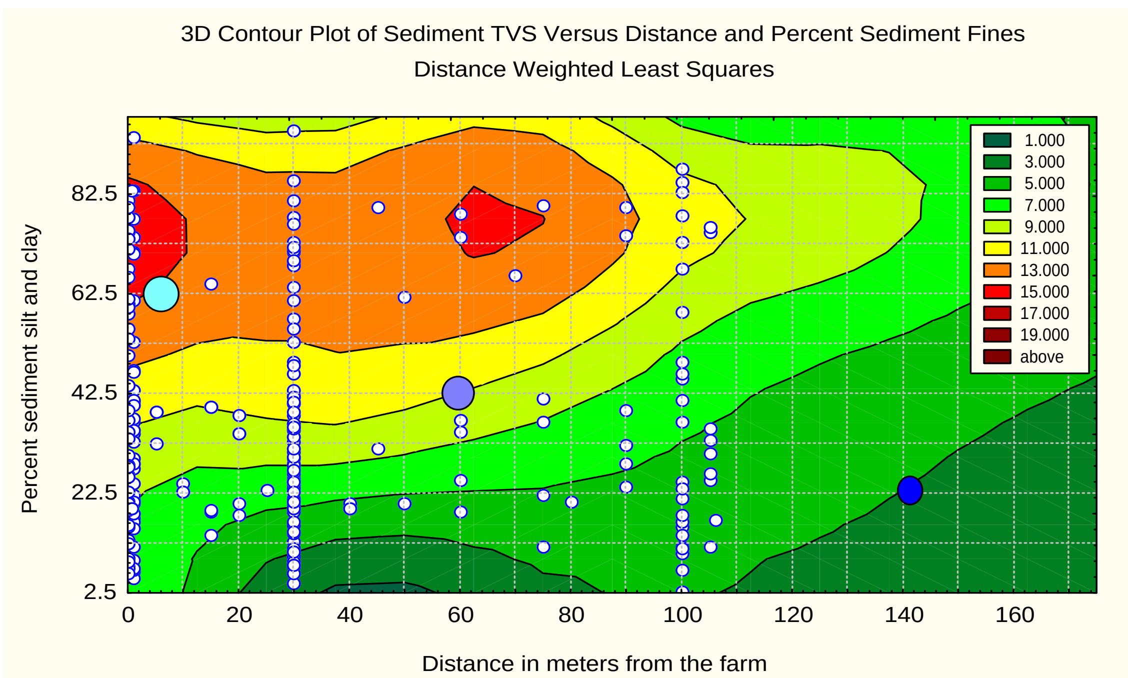

Figure 7. Sediment TVS as a function of distance from the perimeter of British Columbia salmon farms located in environments characterized by different proportions of silt and clay in the sediment grain size distribution. The circles filled in blue represent the upper 90th percentile TVS observed at British Columbia reference stations having 22.5, 42.5 and 62.5 percent sediment silt and clay.

communities can infuse oxygen and sulfate into the top 4 to 6 cm of fine-grained sediments. Oxygen is consumed biologically by prokaryotes and eukaryotes and chemically through chemical oxidation in sediments. In sediments with high organic content, bacterial catabolism of organic materials can create significant BOD along aerobic pathways. When this BOD equals the diffusion and infusion of oxygen from the overlying water column, the sediments are at their assimilative capacity for organic matter. As organic inputs increase further, oxygen levels drop, and the sediments become reducing – leading to the exclusion of some infauna. Therefore, unlike TVS, reduced redox potential affects infaunal communities, regardless the form of TVS.

There is a rich literature describing oxygen uptake in sediments and the resulting redox potential measured using ORP probes and field meters. Measurements of sediment redox potential have been found to be highly variable (Brown *et al.* 1987; Hargrave *et al.* 1993, 1995; Wildish *et al.* 1999), which detracts from their use in regulatory programs (Wildish *et al.* 1999). Henderson and Ross (1995) noted that, "Eh, sulphide and carbon values across the whole study area showed remarkable variation, as other workers have reported and could not be easily used to generalize on the degree of impact." However, GESAMP (1996) lists redox potential as having moderate usage, low cost and high value. Brown *et al.* (1987) observed seasonal trends in sediment redox at salmon farm sites with highest levels reported in February followed by a decline in May and August. Sediment redox was constantly reducing within three meters of the cages (-146 to –186 mV), seasonally reducing at 11 meters from the cages (-185 mV in May) and positive Eh was observed in February and August. Sediment redox was positive at all stations in all seasons at a distance of 15 meters and beyond. Hargrave *et al.* (1993) observed similar seasonal trends with increased oxygen uptake, increased ammonium flux, and increased abundance of *Capitella capitata* during summer months (July through September). Interestingly, there appeared to be a direct relationship between the abundance of *C. capitata* and sediment redox. Pamatmat *et al.* (1973) observed oxygen consumption rates in Puget Sound that ranged from 4 to 56 ml O_2/m^2-hr. Bacteria, meiofauna and infauna accounted for 10 to 50% of this consumption and chemical oxidation accounted for the rest. These authors observed that oxygen uptake in sediments under the Clam Bay salmon farm were significantly higher at 125 ml O_2/m^2-hr. However, the oxygen consumption rates declined significantly with distance and reached reference levels within 30 meters of the farm. Meijer and Avnimelech (1999) used microprobes to examine oxygen tension in sediments and water in organically enriched freshwater fishponds. They found that absent bioturbation, oxygen penetrated the sediments only to a depth of a few millimeters. The calculated oxygen consumption of 45 to 50 mg O_2/m^2-h was related primarily to biological (bacterial) activity. Negative Eh values were reported at all sediment depths > ca. 2.0 mm with high fish production. Redox potential was positive above a sediment depth of 20 mm at low levels of production and sediment redox was positive at all depths less than 30 mm when nitrate was added to the ponds. The other interesting point made in this paper is that even though sediments were highly reducing at depths greater than 1 to 2 mm, the overlying water was essentially oxygen saturated at a height of 1.0 mm above the sediments – emphasizing the independence of oxygen concentrations in the water and in sediments – even when the sediments were anaerobic. A similar conclusion was reached by Cross (1990) who did not observe decreased dissolved oxygen concentrations in bottom water at 7 of 8 farms surveyed when compared with local reference stations. Reduced bottom water dissolved oxygen concentrations of 3.1 to 3.5 mg/L were observed at the eighth farm. In contrast, EVS (2000) reviewed other reports indicating that sediment oxygen demand can lead to depressed oxygen levels in the overlying water (Gowen *et al.* 1991*;* Tsutsumi *et al.* 1991).

In summary, sediment redox potentials are dependent on sediment grain-size distribution, depth of the benthic boundary layer, bioturbation, organic loading and oxygen tension in the overlying water column. A variety of conditions have been observed, but the literature suggests increased oxygen demand and the potential for reducing conditions in sediments within 10 to 15 meters from many (but not all) salmon farms. The literature also suggests that BOD will increase in summer and decrease in winter in enriched sediments. This will result in lower redox potential and increased biological effects in summer and lower responses in winter. The literature also suggests a great deal of variation for redox readings in sediments from a single sample station. No information was obtained that would help partition the variance into instrument, method, technician or true environmental compartments.

Sediment free sulfides ($S^{=}$). Numerous sources of organic carbon contribute to sediment accumulations in coastal waters. These include authochthonous sources like benthic diatoms and dead infaunal organisms and allochthonous sources such as planktonic detritus, drift macroalgae, eelgrass and terrigenous inputs – particularly in forested regions. These organic materials are degraded aerobically on the surface of sediments. However, oxygen penetration in muddy or sandy sediments is typically restricted to the top few millimeters or centimeters (Heij *et al.* 1999 or Wang and Chapman 1999). Below that depth, the oxidation of organic matter rapidly depletes free oxygen and organic matter is oxidized by the reduction of sulfate to sulfide by *Desulfovibrio* and *Desulfotomaculum* bacteria (Kristensen *et al.* 2000). The importance of sulfate reduction should not be underestimated (Luckge *et al.* 1999). Kristensen *et al.* (2000) observed that sulfate reduction rates in the top 10 cm of sediment under netpens accounted for 75 to 118% of the CO_2 flux across the sediment water interface. They also observed that sediment metabolism beneath the netpens (525 to 619 mM CO_2/m^2-d) was ten times higher than at a local control station (24 to 70 mM CO_2/m^2-d). In the absence of sufficient sulfate, further catabolism of organic matter is accomplished by methanogenic bacteria, producing ammonium (NH_4^+) by stripping oxygen from NO_4. Figure 10 is a simplified diagram describing the cycling of sulfur in marine sediments. Other pathways involving organic sulfur have been omitted and only major pathways included. It should be noted that hydrogen sulfide (H_2S) dissociates in water as a function of pH (i.e. $2H_2S \Rightarrow 2HS^- + H_2$). At pH 6.0, 91% of sulfide is in the hydrogen sulfide form. At pH 7.0 this decreases to 50% and at pH = 8.0, typical of seawater, only 9% of sulfide is in the H_2S form (Wang and Chapman 1999). The S^{2-} form readily complexes with iron in seawater (Heijs *et al.* 1999) and has rarely been observed as a dominant free form of sulfur in marine environments (Wang and Chapman 1999).

Chanton *et al.* (1987) observed that the quantity of sulfate reduced by heterotrophic bacteria was greater than the quantity of reduced sulfur buried in the form of iron sulfide or pyrite. That is because much of the total soluble sulfides ($S^{=}$) were oxidized to sulfate in the aerobic zone of the sediments or at the sediment water interface in the presence of the sulfur oxidizing bacterium *Beggiatoa.* One can think of sulfate as a recyclable fuel that drives the engine. The end products of anaerobic metabolism in sediments are buried iron sulfide and pyrite, carbonate, and a variety of forms of soluble sulfur ($S^{=}$) including hydrogen sulfide. These soluble sulfur compounds continue the cycle until either the organic substrate is exhausted or the soluble sulfides are bound by metals and sulfate is exhausted. Dissociated sulfides ($S^{=}$ or HS^-) and hydrogen sulfide (H_2S) comprise most of the soluble sulfides measured using silver/sulfide probes. These soluble forms plus FeS represent the acid volatile sulfide (AVS) portion, and all of this plus pyrite is referred to as chromium reducible sulfur (CRS). Just as it is important to maintain adequate oxygen for aerobic respiration, it is equally necessary to maintain adequate sulfate levels in sediments to sustain the anaerobic pathways described in Figure 10. Once the supply of sulfate is depleted, *Desulfovibrio sp.* bacteria can no longer catabolize complex organic matter and the system shifts to slower methanogenic processes. Therefore, sediment characteristics that enhance the diffusion and/or infusion of seawater will not only sustain aerobic metabolism at high levels of organic input, but they will sustain anaerobic pathways for longer periods of time when the assimilative capacity is exceeded.

Kristensen *et al.* (2000) observed that decreased sulfide concentrations as a function of depth to 15 cm were associated with a lack of carbon substrate and not due to reduced sulfate concentrations. Data in Cranston (1994) from areas with low organic inputs also revealed adequate sulfate concentrations – even in very deep sediments. However, Cranston (1994) also presented data for a site with high organic carbon content where sulfate was depleted and a significant portion of the carbon residue was buried. That is likely why some salmon farms, located in fine-grained sediments, take a long time to remediate. In some environments, both free oxygen and sulfur pathways are overwhelmed by the oxygen demands of first, aerobic organisms, and then of sulfur reducing bacteria. It appears that the top few millimeters are where most of the action occurs with respect to both aerobic and anaerobic catabolism. The colonies of *Beggiatoa* bacteria are a healthy sign in that they are catalyzing the breakdown of underlying organics by efficiently oxidizing sulfide and recycling sulfate back into surficial sediments.

The capacity of the various pathways illustrated in Figure 8 depends on a number of factors including the availability and supply of divalent cations, including iron (Fe^{2+} and Fe^{3+}), plus sulfate and oxygen in sediments. Heijs *et al.* (1999) provides a methodology for partitioning sulfide along some of these pathways. Seawater and marine sediments typically contain sufficient amounts of sulfate to fuel the catabolism of natural organic compounds. Cranston (1994) determined the concentration of ammonium, sulfate and organic carbon as a function of depth in deep (200 meter long) cores from Halifax Harbor. Cores containing low concentrations of organic carbon (<0.4%) also contained significant quantities of sulfate (>10 to 20 mM SO_4) to depths of at least 200 meters. These cores contained small concentrations of ammonium (NH_4^+) suggesting that sulfate reduction was responsible for most of the catabolism. Sediments containing intermediate quantities of organic carbon (0.4% <TOC<2.5%) demonstrated sulfate depletion beginning 20 centimeters below the surface with zero sulfate at 80 meters below the surface. These cores contained up to 2.5 to 3.0 mM of NH_4 indicating the increasing importance of methanogenic pathways in the presence of increasing organic content that depleted the sulfate pool. Lastly, cores containing 5.0 to 6.0 percent organic carbon were depleted of SO_4 below a depth of 20 cm where HN_4^+ concentrations were 2.5 to 6.0 mM. Consistent with this pattern, organic carbon was exhausted at depth in the lightly loaded sediments but persisted at all depths where carbon concentrations were 5 to 6%. These observations are important to the fate of organic carbon at salmon farms where surface sediment accumulations of 25 to 35% percent are not uncommon. At these organic carbon concentrations, particularly in fine-grained sediments, which inhibit the intrusion of oxygen to re-oxidize sulfide to sulfate, the supply of sulfate to fuel *Disulfovibrio* catabolism of organic carbon may become depleted. Under these circumstances, the system would essentially stall. Sulfides would be converted to iron sulfide and pyrite. When the supply of Fe^{2+} and Fe^{3+} ions is exhausted, soluble sulfide in the sediments would essentially remain at a static level and further carbon degradation would occur only along energetically more expensive and therefore slower methanogenic pathways. This hypothesis would explain the long chemical remediation times and persistence of elevated sulfide concentrations at a few farms located over fine-grained sediments (Brooks *et al.*, 2004a). This hypothesis would also suggest that increasing the flow of sulfate and oxygen into these sediments would restart the aerobic and perhaps more importantly the sulfate-sulfide engines resulting in reduced remediation times. This hypothesis should be explored by evaluating the entire sulfur pool in sediments at slowly remediating sites to determine if sulfate is exhausted and if the pool of dissociated Fe^{2+} and Fe^{3+} ions has been depleted.

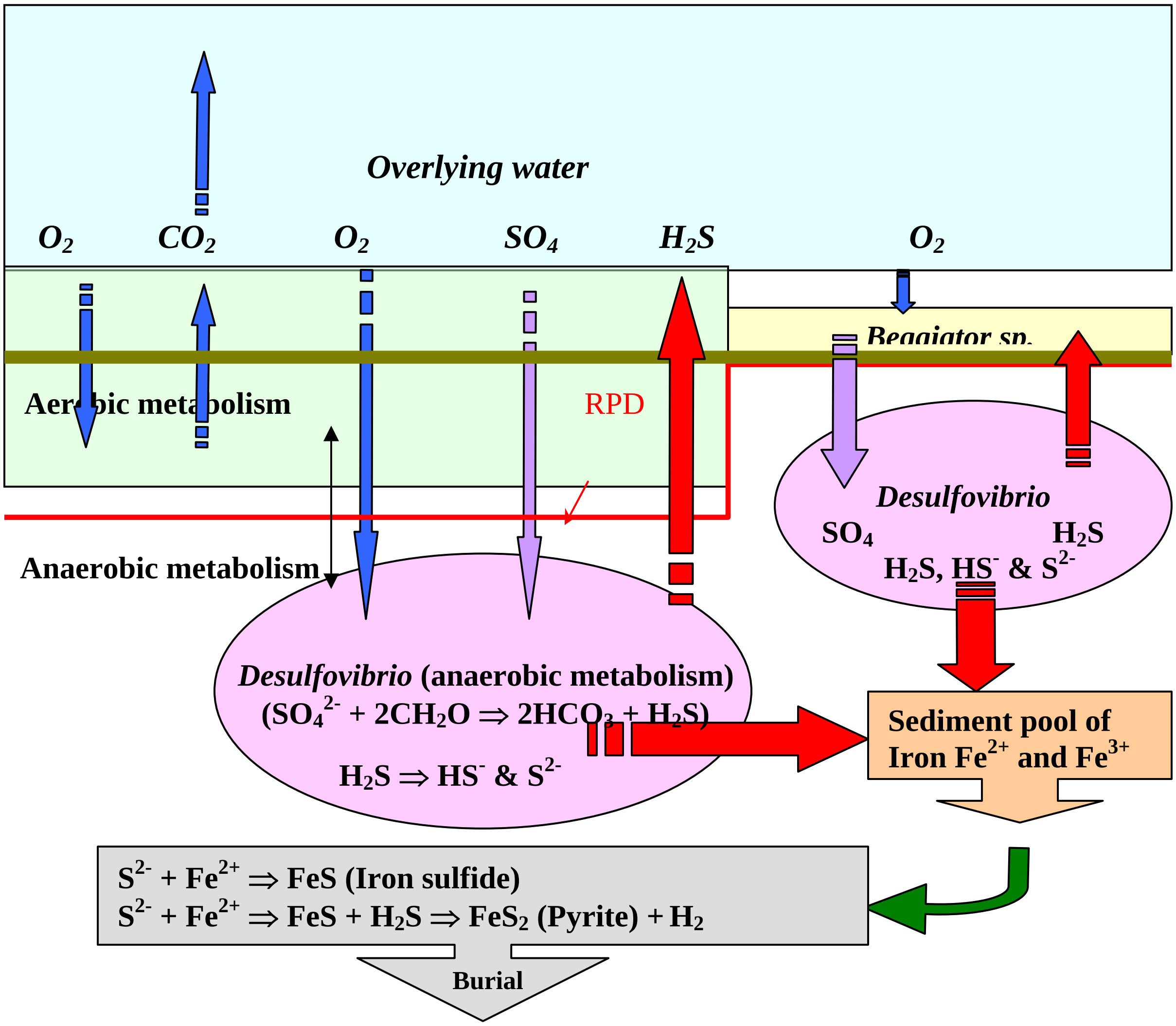

Figure 8. Major sulfur pathways in marine sediments.

Sediment concentrations of total sulfide ($S^{=}$) collected between 1996 and 2000 (Brooks, 2000b) and are summarized in Figure 9. It should be emphasized that the sulfide probes used in collecting this data measure the total soluble sulfide (HS^-, H_2S, and $S^{=}$) available in sediments – they do not measure FeS^- or FeS_2 concentrations. Sulfide concentrations exceeding 4,000 µmoles were restricted to distances $\leq$ 60 meters from the perimeter of netpens. Perhaps of more importance, sediment sulfide concentrations exceeding 600 micromoles were observed as far as 135 meters from the perimeter of netpens. Note that higher sulfide concentrations were observed at greater distances (130 to 140 meters) from farms located in depositional areas characterized by fine-grained sediments, than from farms located in erosional areas. Sediment sulfide data has been reported by Kristensen *et al.* (2000) and Holmer and Kristensen (1992) for sediments close to commercial netpens in the Wadden Sea. Hargrave *et al.* (1995) reported sulfide concentrations at 2.0 cm depth intervals in Bay of Fundy sediments from under salmon farms and at reference stations. All reference station sediments contained > 800 to 1000 µmoles $S^{=}$ at sediment depths >14 cm and more generally at sediment depths >4.0 cm. Surficial sediment (0 to 2.0 cm depth) concentrations of sulfide were less than 280 μmoles. This is consistent with the previous review indicating that anaerobic conditions should be expected in fine-grained reference stations at sediment depths greater than 1.0 to 2.0 cm. It is only the surficial sediments that are typically aerobic.

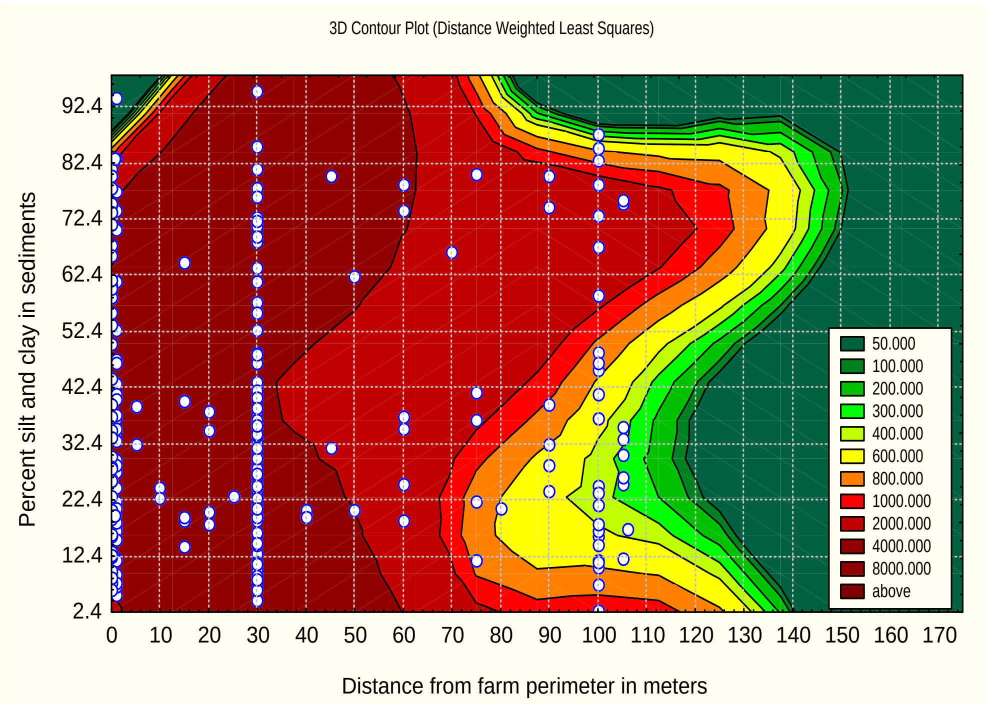

Figure 9. Contour plot describing sediment sulfide concentrations (micromoles) as a function of the percent fines in sediments and distance from the perimeter of farm netpens.

In contrast to reference conditions, Hargrave *et al.* (1995) recorded mean (± 95% confidence interval) surficial sediment sulfide concentrations of 1084 ± 475 with a range of 180 and 4,200 μm adjacent to salmon farms. Wildish *et al.* (1999) reported 1998 surficial (2.0 cm depth) sediment concentrations of sulfide at Bay of Fundy salmon farms that averaged 2.3 times higher at 3,280 ± 472 μmoles with a range of 20 to 36,000 μmoles. The differences may have attributable to different salmon production levels during the two studies or to slight differences in analytical technique. Wildish *et al.* (1999) also report sulfide concentrations in surficial sediments under intensive mussel cultures that averaged 11,476 ± 3,046 μmoles with a range of 180 to 57,000 μmoles and Brooks (2005c) reported sulfide concentrations between 12,800 and 15,300 μM under raft cultured mussels in Washington State. These latter findings support the hypothesis that the intensive culture of all species can result in exceeding the assimilative capacity of local sediments leading to high concentrations of sulfide. The results of Brooks (2001) are generally consistent with those of Wildish *et al.* (1999) and suggest that sediment concentrations of sulfide under and on the perimeter of intensive aquaculture operations vary with production levels and with local bathymetry and hydrodynamics and that they can reach concentrations > 20,000 μmoles. Brooks (2001), Brooks and Mahnken (2003) and Brooks *et al.* (2003c, 2004) are the only reports found in the literature that have examined sediment physicochemical characteristics at distances greater than 100 meters from salmon farms.

Biological response to physicochemical changes in sediments. Brooks (2001) reported that both sulfides and redox potential were well correlated with nearly all endpoints describing macrobenthic communities near salmon tenures and at reference locations in British Columbia. Free sediment sulfides measured immediately in the field were the most reliable predictor of biological effects and subsequently became the focus of the British Columbia Marine Netpen Waste Regulation. Figure 10 describes the log transformed number of taxa observed by Brooks *et al.* (2004) in Carrie Bay sediments. In general, marine macrofauna are sensitive to increases in $S^{=}$ with a lower low effects thresholds of a few tens of micromoles. Free sulfides at reference stations are generally < ca. 350 μM. However, sulfides are elevated where ever there are large accumulations of animals including natural shellfish beds in intertidal environments and around piling, which frequently support large and diverse communities of organisms (Goyette and Brooks 1998, 2000).

In contrast to the diversity of animals, their abundance is frequently increased near fish culture operations (Figure 11). At least eight species of annelids, mollusks and crustaceans have been identified proliferating in enriched sediments. Reference sediments in deep water typically support 50 to 60 types of organisms in an abundance of about 4,500 animals/m^2 (WDOE 1996). Macrofaunal abundance reached 189,000 animals/m^2 at some farms in Clayoquot Sound, British Columbia (Brooks, 2001). The most abundant organism was the crustacean *Nebalia pugettensis,* which has also been found proliferating around piling in Puget Sound (Brooks 2004c). Figures 12 and 13 describe the number of taxa and abundance of macrofauna as a function of distance from the perimeter of a salmon farm in British Columbia located in an area of relatively slow currents. At the time of the study, the farm was producing 1,500 metric tonnes of salmon during each 20 month growout period followed by a six month fallow. There has been no proliferation of any taxa at this farm. The macrobenthic community was dominated by mollusks, which are less likely to proliferate than are annelids.

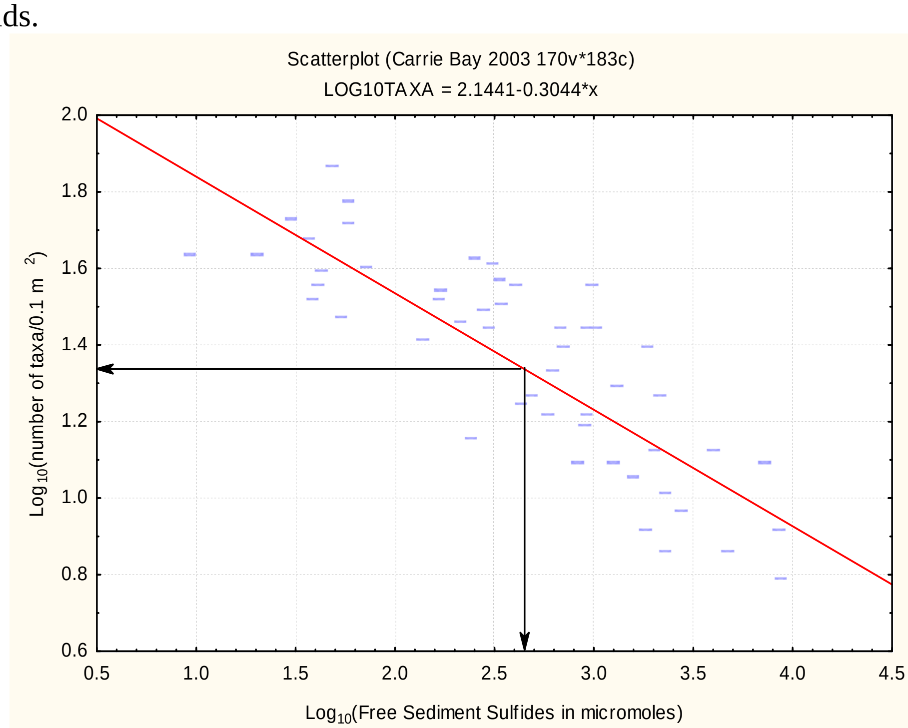

Figure 10. The number of macrofaunal taxa observed in 0.1 m2 van Veen grab samples sieved on 1.0 mm screens at Carrie Bay in 2000, 2001 and 2002. A 50% reduction in the maximum number of taxa observed in these surveys occurred at a sulfide concentration of 447 μM S=.

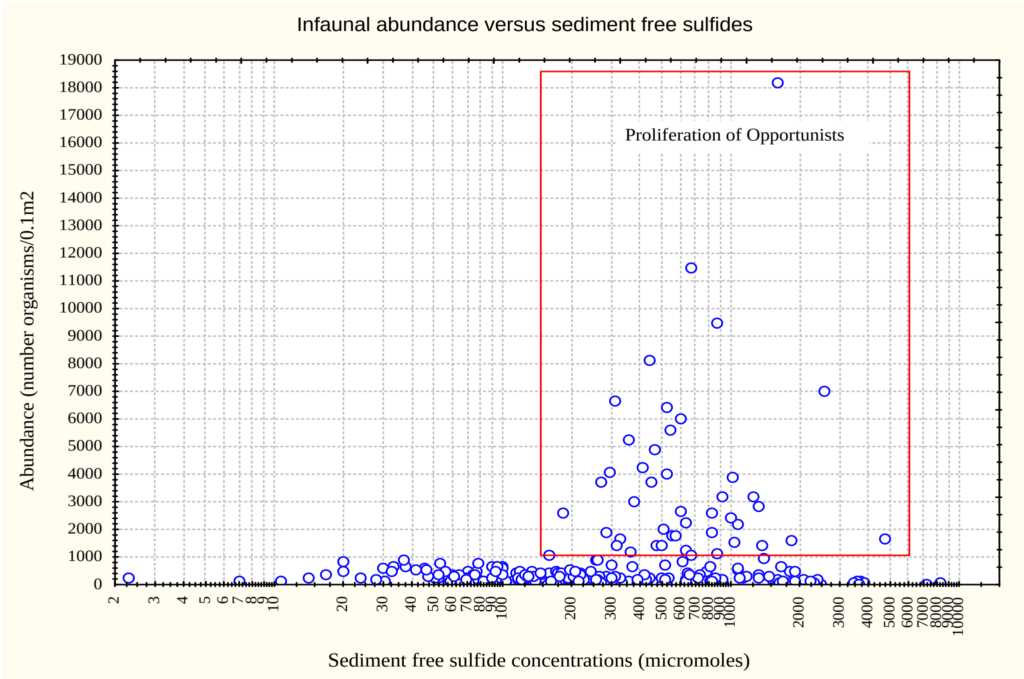

Figure 11. Abundance of macrofauna as a function of free sediment sulfides (μM) at seven British Columbia salmon farms reported in Brooks (2001).

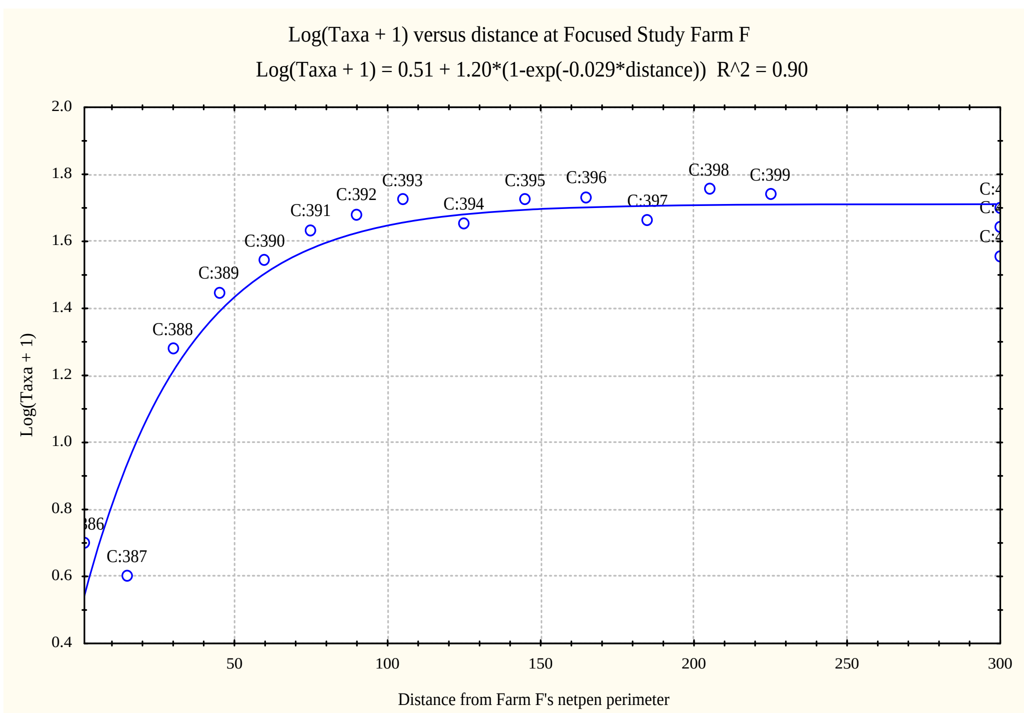

Figure 12. Number of taxa observed in 0.1 m^2 modified van Veen grab samples at the Upper Retreat salmon farm in British Columbia as a function of distance (m) from the netpen's perimeter.

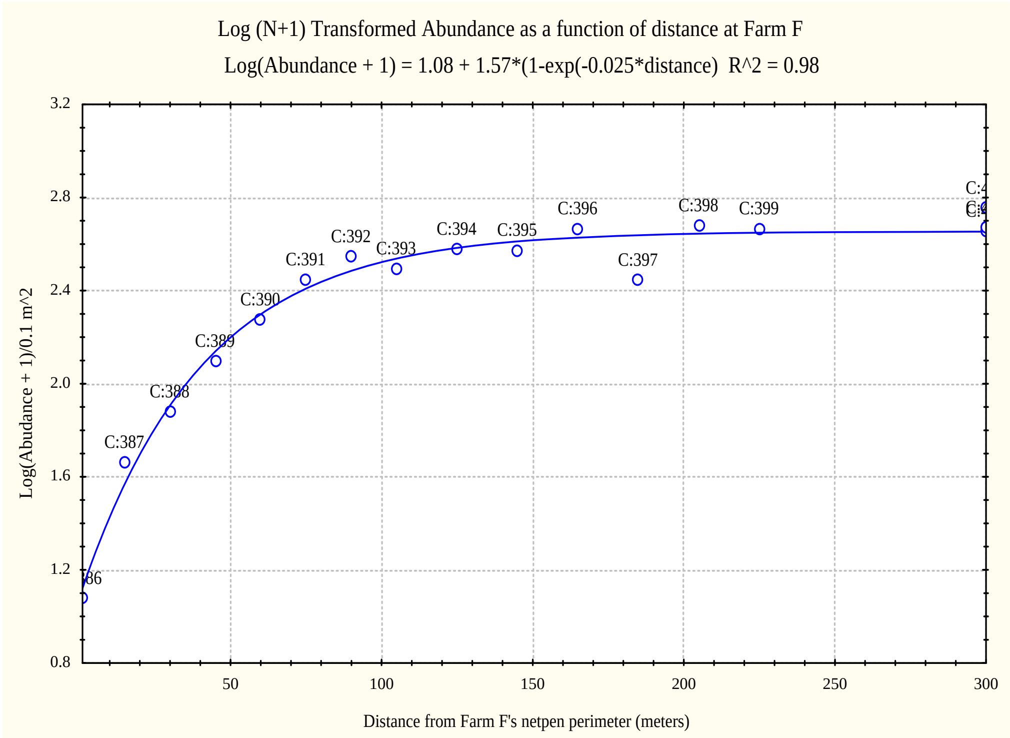

Figure 13. Number of organisms observed in 0.1 m^2 modified van Veen grab samples at the Upper Retreat salmon farm in British Columbia as a function of distance (m) from the netpen's perimeter.

The question arises as to the maximum cultured biomass that can be grown at a site without exceeding the sediment's assimilative capacity for labile organic matter. Figure 14 describes a methodology developed by Brooks (2001) for assessing this question. However, the method is backward looking – not predictive. It requires a comparison of time series of sulfide concentrations and/or redox potentials with fish biomass during a production cycle.

In general, numerous studies of this kind have found that free sulfides increase rapidly during the early stages of production when salmon biomass is still small and feed rates are low. In the case of Upper Retreat, a performance standard allowing 1,000 µM $S^=$ at 100 m distance from the netpen would have restricted production to < 120,000 kg. It should be stated that the author monitors several broodstock holding sites in British Columbia where the maximum biomass is generally <50,000 kg. Detectable effects have rarely been observed in sediments at these sites. The spatial extent and degree of benthic effects do not appear to increase linearly with increasing production. Current production rates in British Columbia have increased to 3,500 to 4,000 mt per farm. Ongoing monitoring has not observed significant increases in the benthic footprint of these farms (Brooks, unpublished). However, chemical remediation times at these higher production levels have not been determined.

The reports cited in this paper generally result from studies of farms representing worst cases where adverse benthic effects have been observed. Brooks (1994b and 1995b) documented sediment chemistry and infauna down current from a salmon farm located in a well-flushed passage in Washington State with maximum current speeds in excess of 125 cm-sec^{-1}. The water was shallow (15 - 18 m MLLW) with sediments dominated by large gravel, cobble and rock mixed with small amounts of sand, silt, clay and broken shell. The site was used for final grow-out as part of a complex, which produced approximately 3,000 mt of Atlantic salmon per year. Monitoring results demonstrated the positive environmental effects associated with this farm, which had been operating continuously for more than 10 years in the same location at the time of the study. A total of 3,953 infaunal organisms distributed in 116 species were observed at the 60 m control station in 1994. The abundance and diversity of benthic infauna was enhanced at all stations closer to the farm with a maximum of 7,350 animals distributed in 173 species observed at the 30 m station. On the periphery of the farm, 4,207 animals were observed, distributed in 142 species. Annelids dominated the infaunal community and *Capitella capitata* (16%) and *Prionospio steenstrupi* (17%) were abundant in the immediate vicinity of the farm. However, arthropods and surprisingly mollusks (*Mysella tumida* and *Macoma* spp.) were well represented in these samples. The abundance and diversity of infaunal organisms was positively correlated with sediment TOC, suggesting that organic carbon was limiting the infaunal community is this area. Significant numbers of fish, shrimp and other megafauna were observed during each annual video survey at this site, which appeared to function as an artificial reef. Three salmon farms located in close proximity to each other all shared the same characteristics. They appeared to attract megafaunal predators and to enhance the infaunal and epifaunal communities.

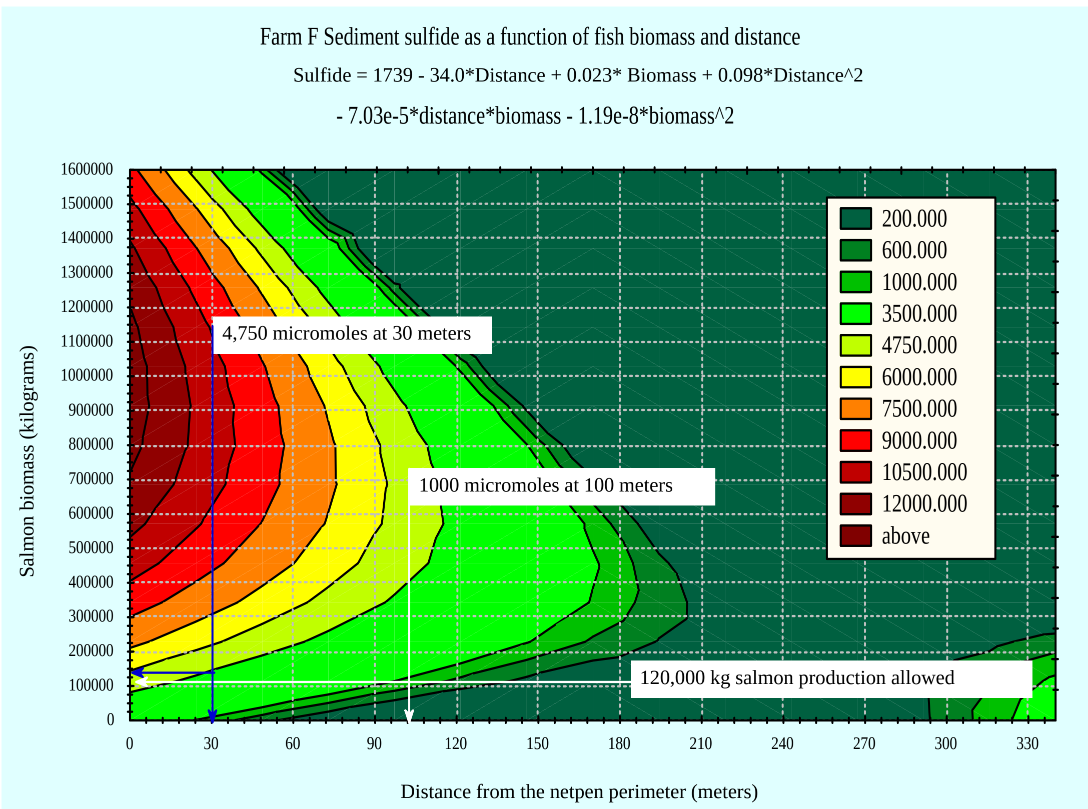

Figure 14. Free sediment sulfide concentrations observed at the Upper Retreat salmon farm in British Columbia as a function of distance from the netpen's perimeter and biomass of salmon in the netpens when the samples were collected.

Changes in the local fish community. Salmon farms are known to function as fish aggregating structures. The structures attract numerous fish species, which frequently take up residence between the containment and predator nets. There are no published reports documenting this community. Brooks (1994b and 1995b) identified large numbers of pile perch (*Rhacochilus vacca*), shiner perch (*Cymatogaster aggregata*), herring (Clupea pallasi), lingcod (*Ophiodon elongatus*), bay pipefish (*Syngnathus leptorhynchus*) and several species of sole (*Pleuronichthys* spp.) at a well-flushed net-pen site in Washington. At another site nearby, located over a sandy bottom, sea cucumbers (*Parastichopus californicus*) and geoducks (*Panopea abrupta*) had proliferated. All of these populations were closely associated with the farms (within 30 m). It should be added that one of these facilities is located in shallow water (15 - 18 m MLLW) and fast currents (115 cm/sec). The second facility is located in a moderately well flushed environment with maximum currents of 60 cm/sec and water depths of 22 - 30 m MLLW.

Chemical and biological remediation of sediments. Chemical and biological recovery of sediments under salmon farms is well documented in the literature by, *inter alia*, Ritz *et al.* (1989), Anderson (1992), Mahnken (1993), Brooks (1993a), Brooks (1999), Brooks *et al.* (2003c, 2004a), Lu and Wu (1998), Karakassis *et al.* (1999) and Crema *et al.* (2000). Brooks *et al.* (2003c) have defined chemical and biological remediation as follows:

Chemical remediation. Chemical remediation is the reduction of accumulated organic carbon under and adjacent to salmon farms to a level at which aerobic organisms can recruit into the area. It appears that initially high levels of sedimented organic carbon decline exponentially and approach baseline conditions asymptotically. Chemical remediation is accomplished through chemical, biological and physical processes.

Biological remediation. Biological remediation is defined as the restructuring of the infaunal community to include those taxa representing $\geq$ 1% of the abundance observed at a local reference station. Recruitment of rare species representing < 1% of the reference area abundance is not considered necessary for biological remediation to be considered complete.

At two sites where long-term fallow studies were conducted by Brooks (2000) and Brooks *et al.* (2004), sediment concentrations of volatile solids declined rapidly as soon as harvests were started and reference physicochemical conditions were achieved within four to six months of fallow. Remediation at the Arrow Pass farm can be inferred from the temporal series of TVS curves in Figure 6. Figure 15 describes the temporal and spatial history of free sediment sulfides at the Upper Retreat salmon farm where chemical remediation was considered complete in 4 to 6 months. Brooks (unpublished) has continued to monitor the Upper Retreat salmon farm during an extended fallow period and it appears that biological remediation was complete after approximately 15 to 18 months of total fallow (six months for chemical remediation and 9 to 12 months for biological remediation).

Not all Northeast Pacific salmon farms remediate this quickly. Carrie Bay was a salmon farm that appeared to create more extensive and dramatic benthic effects than any other site in the Broughton Archipelago of British Columbia. As soon as the extent and degree of the benthic impacts became known to management, they terminated operations there and the site was voluntarily studied during a seven year fallow period. This farm was located in a highly depositional area and benthic conditions were exacerbated by poor feeding practices. Brooks *et al.* (2004a) found that chemical remediation was nearing completion but was not yet complete after five years in fallow. The sulfide history at this site during the fallow period is described in Figure 16. Chemical remediation was proceeding steadily, but was not complete in 2002 following five years in fallow. Regression analysis suggested that seven years would pass before sediment chemistry at this site returned to baseline conditions.

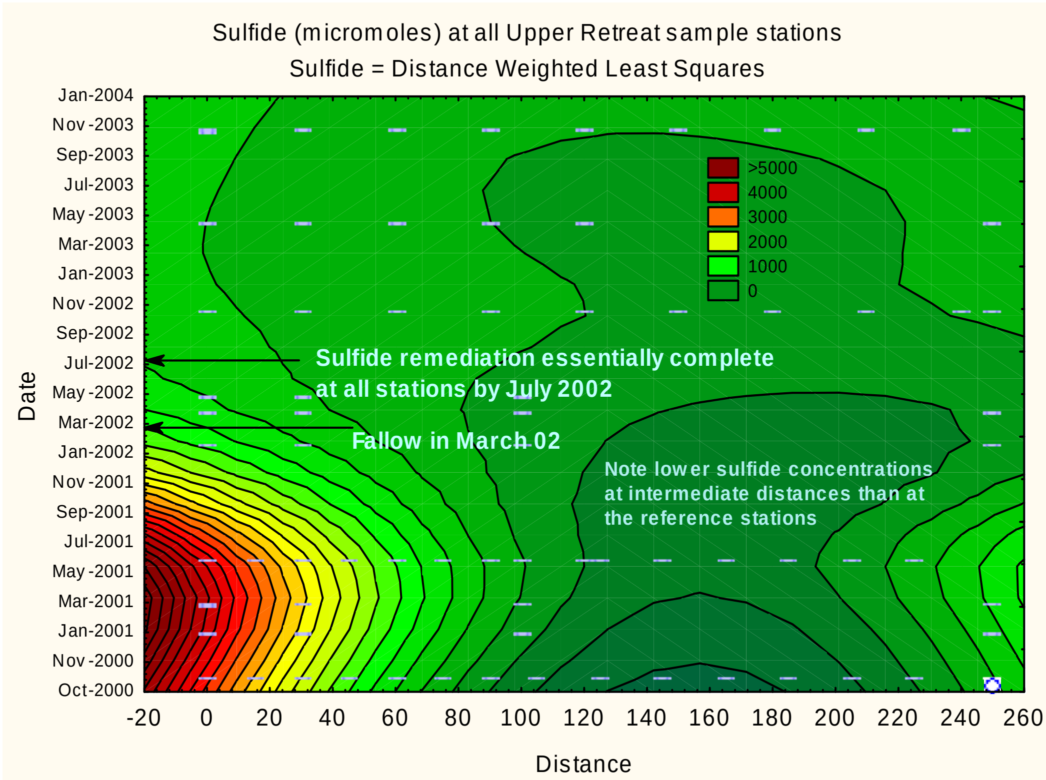

Figure 15. Free sediment sulfide history for all transects as a function of date and distance from Upper Retreat's netpen perimeter.

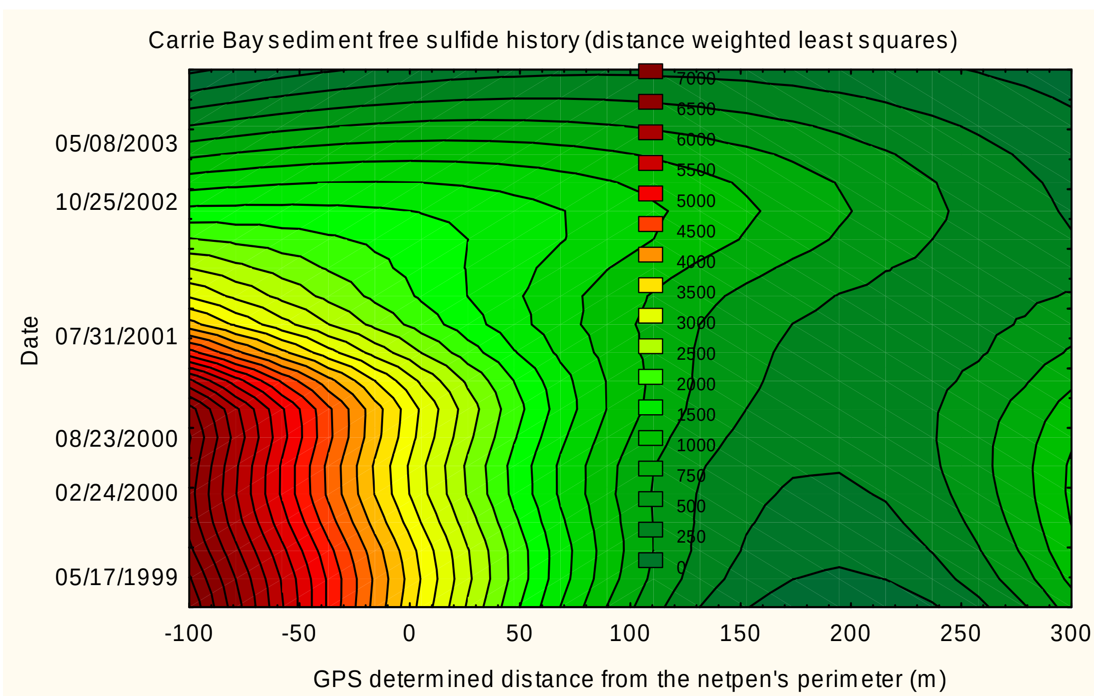

Figure 16. Distance weighted least squares contour plot describing concentrations of free sediment sulfides near the Carrie Bay salmon farm between 1999 and 2003.

Future studies may extend the range of times required for chemical and biological remediation. However, at present, it appears that most salmon farm sites chemically remediate in six months to a year in the Northeast Pacific and that biological remediation, as defined above, occurs during the next invertebrate recruiting season, which is a year or less depending on the season when chemical remediation is complete.

Assessing the environmental costs associated with benthic effects near salmon farms. Brooks (2001) estimated the lost fin-fish production associated with the diminished macro-invertebrate biomass within the footprint of seven salmon farms in the Broughton Archipelago. Macroinvertebrate wet tissues were weighed on a four place balance as part of the community inventories at these sites. The biomass observed at the local reference station was assumed to have been diminished within the average footprint observed at salmon farms (i.e. an area of 1.6 hectares where sulfide concentrations exceeded 4,000 μM). This biomass was assumed to replicate itself once per year and it was assumed that all of this production was consumed by a food fish at the next higher trophic level with an efficiency of 0.10. The loss of wild fish was most heavily influenced by benthic productivity at the reference station, which varied by a factor of approximately 6. Between 32 and 1,475 kg of wild fish production were predicted to be lost at these sites where between 175,010 and 1,800,000 kg of Atlantic salmon were present when the surveys were completed. The ratios of cultured salmon to lost wild fish production varied between ca. 1,000 and 34,000 (Table 4).

Table 4. Production of Atlantic salmon and estimated loss of wild fish due to reductions in the benthic invertebrate community biomass at salmon farms described in Brooks (2001)

Farm	Reference Station Biomass (kg macrofauna/1.6 Ha)	Wild fish lost (kg)	Salmon produced (kg)	Ratio Cultured.Wild
A	300.4	110	175,010	1,589
B	172.4	59	650,000	11,024
C	121.0	38	1,100,000	28,646
D	106.0	32	1,100,000	33,951
E	543.8	311	1,800,000	1,475
F	611.7	1475	1,425,153	966
G	333.3	123	1,321,627	10,717

Summary. A detailed description of the nearfield benthic effects associated with salmon aquaculture in the Northeast Pacific has been presented to assess a portion of the environmental costs associated with this form of food production. From an overall perspective, the results presented herein suggest that there was an average loss in production of 306.9 ± 484.5 kg of wild fish at these farms where an average of 1,081,684 ± 492,374 kg of salmon was present at the time of the surveys. The production of Atlantic salmon was, on average, 12,624 ± 12,521 times greater than the lost biomass of wild fish. The marine grow-out phase lasts approximately 18 to 20 months and adding another 24 months for chemical and biological remediation suggests that the sediments were negatively affected for 44 months. Several conservative assumptions (from the environment's point of view) were necessary to define these costs and the actual loss of wild production in the near field will likely be less, on average, than 307 kg of fish during a complete production and fallow period lasting 44 months (84 kg/year for 3.7 years).

This analysis accounts only for the near-field effects of enrichment. Brooks (2001) did not detect either physicochemical or biological effects at distances >205 meters from any British Columbia salmon farm. However, as the intensity of fed aquaculture within an ecosystem increases, the potential for small, but cumulative, effects from several farms may change natural productivity in the far-field. These far-field effects are difficult or impossible to detect using point in time surveys. Detection requires long-term monitoring to establish trends. Management of cumulative effects requires inventories of all of the contributors to the effect and different management techniques, such as Total Maximum Daily Loading (TMDL) approaches. Far field effects can be serious and need to be avoided. Computer modeling may provide the best approach to determining the assimilative capacity of an ecosystem and this information is necessary to manage the overall scale of aquaculture At present, far-field effects have not been observed at the relatively low density of netpen operations in the Pacific Northeast. They are therefore a Category IV hazard and a quantitative environmental cost assessment is not possible at this time.

4.0. Putting the landscape (footprint) costs of salmon production in perspective with the costs associated with other forms of food production. Assessing the environmental costs of other food producing activities is being undertaken by other contributors in these proceedings. However, the following comments are provided in an attempt to put the costs of salmon aquaculture into perspective with the environmental costs of producing an equivalent amount of beef.

Beef cattle production. Figure 17 is a photograph of an old growth forest in the Canadian Rockies. These forests and their associated wetlands support small, but diverse, communities of plants and animals. The organic debris created by wind-thrown old-growth cedar, Douglas fir, true firs, hemlocks and birch trees creates a dense detrital food web that support marvelous communities of fungi, ferns, mosses and lichens. Many of the Douglas fir trees are five and six feet in diameter. They do not have a limb on them for perhaps the first hundred feet of their 200 foot heights and they are (by actual tree-ring counts) several hundred years old. The creation of such a forest takes centuries, if not eons.

Figure 17. Old growth forest on Horsefly Lake in the Canadian Rockies.

Figure 18 describes a beef cattle farm on the Olympic Peninsula in Washington State, which was once home to a similar forest. Its remnants are seen in a few mature Douglas fir trees and in the eight to twelve foot diameter cedar stumps left from the original logging, which occurred in the middle of the 19^{th} century. Today, about half of the farm has been replanted to Douglas fir and half remains as pasture for Angus beef cattle (Figure 19). The hanging weight of a Black Angus steer is about 70% of its live weight and rendering the carcass into edible meat further reduces the yield to about 42% of the animal's live weight. Gutted and bled Atlantic salmon represent 84% of their live weight. Assuming that the heads are not consumed, the yield of salmon filets is approximately 50% of the live weight (Gary Robinson, Marine Harvest, personal communication). Therefore a salmon farm producing 2,500 mt of live salmon would supply 1,250 metric tonnes of edible filets which are equivalent to 5,411 steers weighing 550 kg each. In the Pacific Northwest, one acre of actively managed pastureland will support one cow for 7.5 months (7.5 animal month units or AMUs). It takes approximately 30 months (30 AMU) to produce a marketable steer and the 5,411 steers require 162,338 AMU or 8,658 acres (3,504 hectares) for 2.5 years. As noted earlier, the benthos under well sited salmon farms chemically remediates in six months to a year and biologically remediates in another year. In contrast, in the Pacific Northwest, it will take hundreds or a thousand years for the pastures seen in Figure 18 to remediate back to the original old growth forest seen in Figure 17.

Figure 18. Whispering Ridge farm on the Olympic Peninsula in Washington State, which was once covered with old growth forests.

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Figure 19. Black angus cattle grazing on pasture that was once old growth forest, but which was removed in an effort to feed Washington State's growing population in the later part of the 19th and early part of the 20th centuries.

Table 5 compares the near field land use costs associated with raising equivalent amounts of edible beef and Atlantic salmon. The table does not assess the possible water column eutrophication associated with tonnes of fish and cattle waste that enters aquatic environments each year. Nor does it assess the ammonium released to the atmosphere, contributing to global warming, or the differences in oxygen resulting from photosynthesis of a mature old growth forest in comparison with pastures. A meaningful life cycle analysis that considers all of the environmental costs associated with both forms of food production would have to be accomplished with the same rigor provided herein for near-field effects and it would span volumes. That is beyond the scope of this report. However, this limited assessment suggests that the landscape directly affected for cattle production is several hundred times greater than it is for production of the same amount of food in salmon aquaculture.

Table 5. Comparisons of the physical footprints associated with production of 1,250 metric tonnes of the edible portions of Atlantic salmon or beef cattle.

Type of food	Edible portion	Live weight	Yield	Spatial footprint	Remediation time
Atlantic salmon	1,250,000 kg	2,500,000 kg	0.50	1.6 hectares	2 years
Angus beef cattle	1,250,000 kg	2,976,190 kg	0.42	6,982 hectares	200 plus years

Harvesting of the ocean's natural bounty. An accurate assessment of the environmental costs associated with recreational and commercial fishing must take into account not only the physical destruction associated with bottom trawling and the poorly accounted for bicatch that is discarded. It must include, among other factors, an accounting of the costs associated with lost production to derelict fishing gear. In 2004, a group of Washington State sport fishermen used side-scanning sonar to identify over 2,000 derelict (lost) shrimp and crab pots in three embayments on the North Olympic Peninsula (Port Angeles Harbor, Sequim Bay and Discovery Bay). They were able to successfully retrieve 292 of these pots. Anecdotal evidence suggests that 40% of the pots were not equipped with sacrificial closure devices designed to deteriorate in relatively short periods to stop long-term entrapment of sea-life. Figure 20 describes the contents of just one of these pots and many similar photographs are available.

Figure 20. One of over 2,000 lost prawn and crab pots identified using side-scanning sonar in three embayments along the Straits of Juan de Fuca in Washington State. As of 2004, 292 of these derelict traps had been retrieved.

Using grant money from Washington State, the fishermen obtained the services of a larger vessel and were able to retrieve masses of lost trawl and gill nets (Figure 21). Lost fishing gear is a world-wide problem that has not been quantified or effectively managed by any jurisdiction that the author is aware of. The recreational fishermen responsible for the program described here commented that the Department of Fish and Wildlife estimated that the several thousand lost pots were killing approximately 10% of the allowable prawn and crab harvests. The same problem occurs in other areas. On two occasions, the author's underwater camera has become entangled in masses of prawn and crab pots snagged on reefs and sunken logs in water depths of several hundred feet.

Figure 21. Mass of derelict fishing nets retrieved from the Straits of Juan de Fuca in Washington State containing hundreds of kilograms of dead and dying fish.

The point in this discussion is not to decry cattle farming or commercial and recreational fishing. The point is to put a portion of the environmental costs associated with Atlantic salmon production in perspective with the environmental costs associated with these more traditional ways of producing food and to assert that the path to sustainability requires fixing the tough problems first and then moving down the scale of effects to fine-tune food production in an effort to achieve true sustainability.

4.0 Sea lice and pink salmon fry. Sea lice of several species naturally infect all salmon and lice are ubiquitous in temperate latitudes. Sea lice have been a significant problem for Atlantic salmon producers in Scotland, Ireland and Norway (Brandal and Egidius, 1979) costing producers £15 to 30 million in Scotland each year (Pike and Wadsworth, 1999). In contrast, sea lice have not been a problem for producers on the Pacific Coast until 2001, when assertions that sea lice originating on farmed salmon were infecting juvenile pink salmon to a degree that threatened the viability of Broughton Archipelago stocks of these fish (Morton and Routledge, 2005). In a precautionary response, the Province of British Columbia has initiated an aggressive *Sea Lice Action Plan* to monitor and control sea lice on farmed salmon and significant research has been undertaken by the Canadian Department of Fisheries and Oceans to understand this issue. Their research is improving our understanding. The following points should are emphasized:

- Sea lice have been observed on juvenile salmon prior to the initiation of salmon farming in the Pacific Northwest. However the prevalence and abundance of these infections was not well documented and there is no baseline data against which to compare current infection rates in the Broughton Archipelago. In contrast, the abundance, prevalence and intensity of sea lice infections on adult salmon is well documented and known to be particularly high on pink salmon returning to natal streams for spawning in the Pacific Northwest (Beamish *et al.*, 2005).

- Sea lice nauplii do not develop to the copepodid stage at ≤ 25 PSU salinity (Johnson and Albright (1991). However, the salinities tested were 15, 20, 25 and 30 PSU. Salinity in Knight Inlet of the Broughton Archipelago varies between 25 and 30 PSU during the period March to June when pink salmon fry are migrating through the estuary (Figure 22). Brooks and Stucchi (2006) used the data of Johnson and Albright (1991) to develop possible response curves for sea lice mortality between 25 and 30 PSU. However, there is a need for rigorous sea lice nauplii mortality within this salinity range.

- As seen in Figure 22, salinity in Knight Inlet (Broughton Archipelago), is generally ≤ 25 PSU from the end of May until early December. Brooks (2005) has hypothesized that this reduced salinity provides a natural control on sea lice during most summers. The salinity data and sea lice abundance on the Humphrey Rock salmon farm support this hypothesis (Figure 23 from Brooks, 2005). Saksida (unpublished) found a similar pattern at this farm. However, Brooks (2005) also showed that salinity patterns within the archipelago are variable from year to year and from one area to another within the estuary.

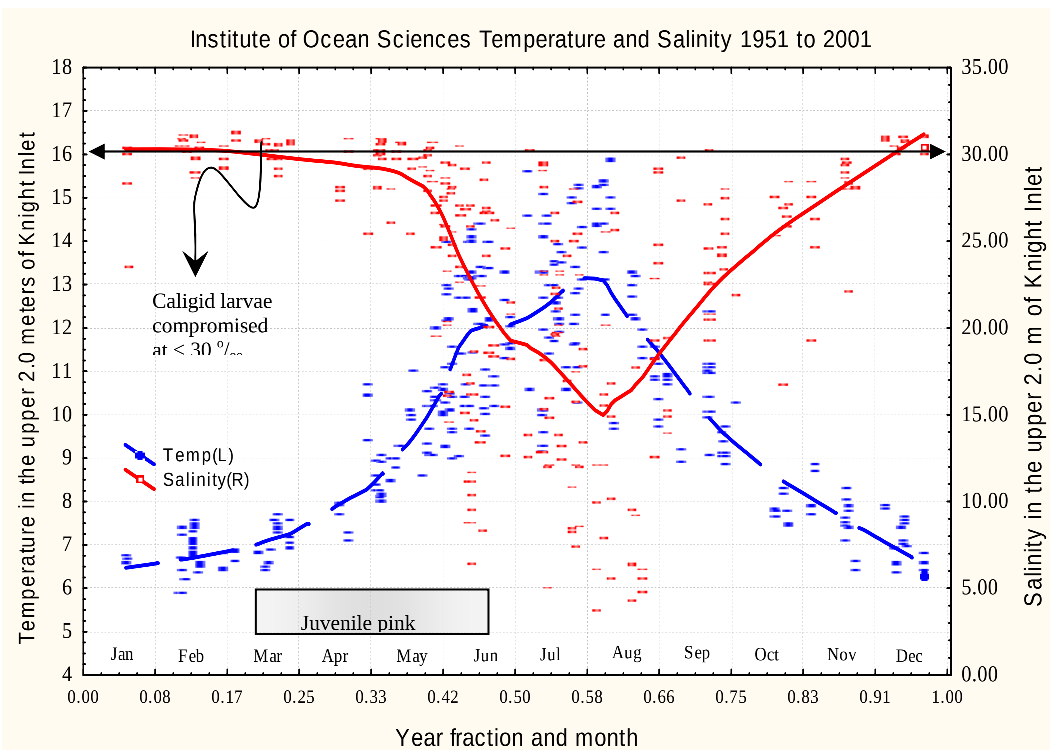

Figure 22. Surface salinity and temperature at depths ≤ 2.0 m in Knight Inlet, Broughton Archipelago between 1951 and 2001 (IOS, 2004). Lowess smoothing creates a local regression model fit for each datum and points close to it; providing a clearer picture of the overall shape of the relationship between the x and y variables.

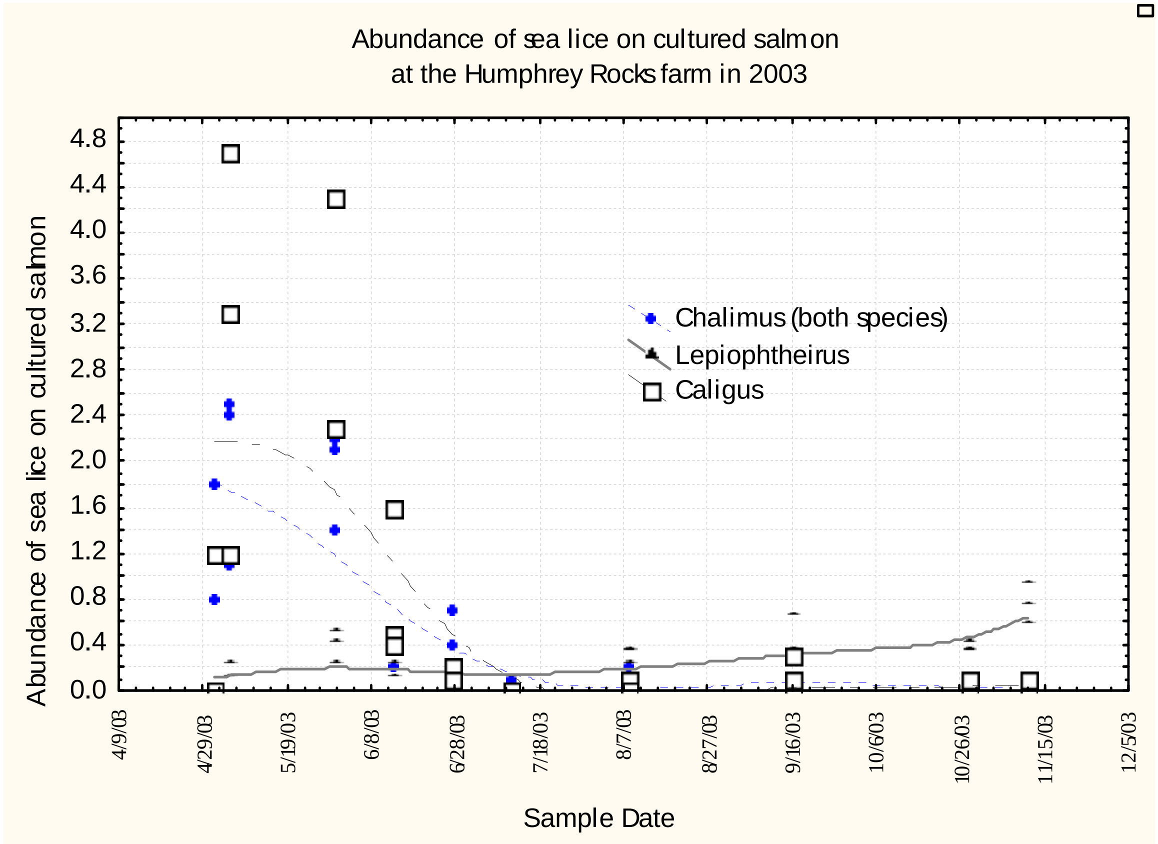

Figure 23. Abundance (mean number lice/fish) of sea lice on cultured Atlantic salmon at the Humphrey Rocks salmon farm located in Tribune Channel, British Columbia. Data from Stolt Sea Farm (2004).

Between 1991 and 2000, pink salmon returns to the Broughton Archipelago increased as the production of cultured Atlantic salmon also increased. In 2000, a record 3.6 million adult pink salmon returned to archipelago watersheds. However, in 2002, the next even year-class declined to 123,000 returns (Figure 24). The increased pink salmon returns during the 1990s were likely associated with operation of artificial spawning channels in Glendale Creek and the Kakweiken River since the late 1980s. Glendale Creek now produces over half of the pink originating in Broughton Archipelago watersheds (Beamish, personal communication). Brooks (2003d, 2005c) has suggested a number of alternative hypotheses explaining the low returns of 2002 including reduced water quality in Glendale Creek associated with the historic 2000 return and depletion of food resources in the Archipelago by an overabundant cohort of outmigration of fry in the spring of 2001. The hypothesis that fry survival is inversely proportional to fry production when adult returns to the Broughton Archipelago exceed 1,000,000 fish has been supported by Beamish *et al.* (2006) and Williams *et al.*, 2003). In addition, lack of food may have made these fry more susceptible to attack by sea lice. The fry, compromised by a number of factors including poor condition and sea lice infections, would have been easier to capture with dip-nets resulting in an overestimation of the abundance and intensity of infections in the general population of pink salmon fry reported by Morton *et al.* (2004).

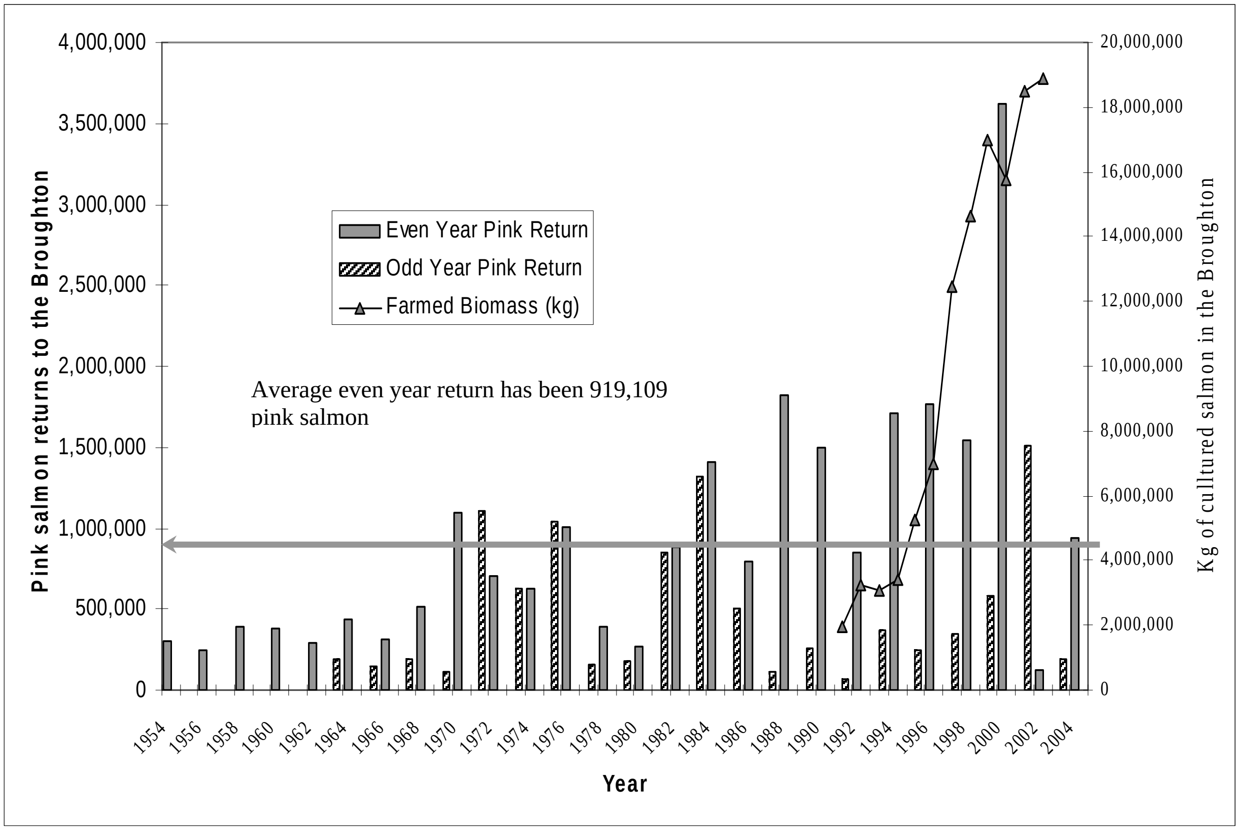

Figure 24. Odd and even year pink salmon returns (vertical bars) and Atlantic salmon production (line) in the Broughton Archipelago from 1962 through 2002.

It has also been hypothesized that the 2002 declines were due to heavy infection of pink salmon fry by sea lice as they migrated to sea in the spring of 2001 and that those infections could only have originated on Atlantic salmon because no other hosts for *Lepeophtheirus salmonis* were present in the archipelago during the spring outmigration of pink and chum salmon fry. Krokosek *et al.* (2005) asserted that pink salmon fry were infected with sea lice in the immediate vicinity of salmon farms. However, Brooks (2005) pointed out that it takes 4.2 to 5.8 days for sea lice nauplii to develop to the infective copepodid stage at spring temperatures of 7 to 9 ^{o}C. The Broughton Archipelago is a very dynamic estuarine system and an analysis of resting current vectors at 15 salmon farms in the archipelago indicated that the nauplii would be transported, on average, 7.3 to 10.0 km from the farms before they molted to the infective stage. Stucchi *et al.* (2005) used a hydrodynamic model for the Broughton Archipelago to predict the location of nauplii when they molt to the copepodid stage and predicted that zones of infection would occur at distances of 10 to 40 km from the evaluated farms. However, their model did not include the effects of wind that would likely affect surface water movements and the dispersion of sea lice. Costelloe *et al.* (1996) concluded that sea lice nauplii hatched at a local salmon farm were swept out of the harbor into the open ocean prior to hatching to an infective stage. Brooks and Stucchi (2006) have advanced several causes for these differing results. Most importantly, Krkosek *et al.* (2005) failed to include the life history of sealice as has been accomplished by Stein *et al.* (2005). This flaw in their model has specifically been noted by other authors including most recently the report of Gillibrand and Willis (2007) whose more complete model predicted that nauplii molted to the infective stage at distances of 7 to 12 km seaward of their source in the modeled loch. The authors noted that their model predictions were consistent with empirical evidence reported by McKibben and Hay (2004). These differing opinions and the importance of preserving stocks of wild salmon have led to five years of intensive research by government, academic and private scientists. The results of that research have revealed the following:

- At least two species of sea lice infect pink salmon fry in the Broughton Archipelago. *Lepeophtheirus salmonis*, previously thought to be specific to salmonids has now been documented in very high intensity on three-spine sticklebacks (*Gasterosteus aculeatus*) by Jones *et al.* (2006b) and a non-host specific louse (*Calligus clemensi*) has been found infecting pink and chum salmon fry during their spring migration. While mature *L. salmonis* has not been observed on sticklebacks, these very abundant euryhaline fish are found sympatrically with pink and chum salmon fry in nearshore environments. Motile sea lice are known to transfer from one host to another in laboratory studies and it is reasonable to hypothesize that sticklebacks provide an over wintering alternate host that allows sea lice returning with adult pink salmon in the fall to infect juveniles the next spring. In addition to this expanded understanding of the hosts of both species of lice, it is known that resident salmonids are present in the archipelago year around – providing additional sources of sea lice larvae. These findings suggest that there are several possible disease vectors responsible for sea lice infections of pink and chum salmon fry. Sea lice monitoring on farmed salmon enables quantification of this source. However, the relative contribution of copepodids originating on farmed salmon will remain unknown until these other disease vectors are better understood.

- The reports described above suggest that Krkosek *et al.'s* (2005) observation of higher intensities of sea lice immediately adjacent to salmon farms is either an artifact of the collection methods or an indication that the source of the infective lice was further inland (upcurrent) in the estuary.

- In the Broughton Archipelago, accumulating data indicates that sea lice infections increase in years when surface seawater temperatures are higher and that the abundance and intensity of sea lice infections is dependent, in part, on salinity. In DFO surveys, sea lice were absent or had a consistently low abundance in those zones where surface salinity was lowest. Sea lice infections have been found to increase near outer areas of the archipelago where salinity is higher and where the fry have been in saltwater for a longer period of time.

- Surface salinity in the eastern portions of the archipelago is significantly reduced by estuarine circulation to <30 practical salinity units (PSU) in late spring and summer. The pre-infective larval stages of *Lepeophtheirus salmonis* have been found to not develop to an infective copepodid stage at salinities <30 PSU. However, the next lowest salinity tested was 25 PSU and data is not available at salinities between 25 and <30 PSU in Johnson and Albright (1991). In addition, no similar studies have been accomplished with *Caligus clemensi.*

- British Columbia's Sea Lice Action Plan requires monitoring of cultured fish and control of sea lice using Slice™ when motile lice abundance reaches prescribed benchmarks. There is now a four year record of the abundance, prevalence and intensity of sea lice on cultured fish demonstrating that infections are lowest when salinity is low and that the program has reduced the prevalence and intensity of gravid female lice on cultured fish to very low levels during late winter and early spring.

- Morton and Routledge (2005) reported high mortality in naturally infected pink salmon held in captivity in comparison with uninfected fish. However, the cause of death was not determined and it can be hypothesized that the infected fish were compromised by some factor (genetics, starvation, underlying disease, etc.) making them more susceptible to sea lice. In contrast, carefully controlled laboratory research by Jones *et al.*, 2006 observed no mortality of infected fish and field surveys by Jones and Nemec (2004) and Beamish *et al.* (2005) have failed to detect a significant detrimental effect on pink salmon fry naturally infected with sea lice.

- Pink salmon returns have varied significantly from year to year since at least 1954, decades before there was any marine aquaculture in British Columbia. These returns continue to fluctuate, but there is no evidence that returns to Atlantic salmon producing areas are significantly affected by the presence of salmon farms.

Sea lice summary. Studies to date indicate that cultured Atlantic salmon are a controllable source of sea lice larvae in the Broughton Archipelago. However, the relative contribution of lice larvae from salmon farms in comparison with natural sources has not been determined because the population of sea lice on wild hosts is unknown. In addition, studies of the effects of sea lice infections on juvenile pink and chum salmon contradict each other. However, evidence from field studies does not appear to support significant mortality of fry associated with observed levels of infection. It should be noted that more recent field studies have been conducted in the presence of an aggressive government program designed to minimize the contribution salmon farms might make to the overall lice population in the archipelago. Perhaps most importantly, returns of pink salmon to the archipelago remain variable within the range observed prior to the advent of salmon farming and there is no evidence that these stocks are in jeopardy of extinction. Beamish *et al.* (2006) concluded that wild Pacific salmon can coexist successfully with cultured Atlantic salmon in the archipelago and Jones *et al.* (2006) summarized the accumulating evidence regarding this issue by noting that,

"We cannot conclude that salmon aquaculture does not contribute *L. salmonis* into the local ecosystem. However, the accumulated evidence suggests that there are important natural sources of infestation and that local factors such as salinity and size of the salmon host in addition to the possible effects of salmon farms that must be considered in future efforts to examine the epidemiology of these parasites."

There are numerous factors affecting populations of Pacific salmon and unequivocal answers to questions like whether or not sea lice from salmon farms are significantly affecting returns in the Broughton Archipelago are difficult to obtain. However, rigorous scientific studies are answering parts of the puzzle and the sea lice issue that was originally touted as a pending crisis has now become simply an interesting scientific question that is slowly being answered.

5.0. Comments regarding recommendation of the NOSB in their Interim Final Report of the Aquaculture Working Group (2006). With respect to the recommendations of the NOSB regarding organic certification of fish produced in open netpens, the author notes that the known environmental costs associated with aquaculture are generally less than those associated with other forms of food production; including terrestrial wheat farming supporting production of a loaf of bread and commercial or recreational fishing harvesting wild stocks of fish. The point being that there are environmental costs associated with al forms of food production and the only rational approach is to manage these costs. Most jurisdictions where salmon aquaculture is practiced have developed waste management policies designed to insure that benthic effects remain localized and ephemeral. Other regulatory programs are designed to minimize the number of escapes and to manage and minimize sea lice originating on cultured salmon. The author objects to the provisions of 205.255 (f) because site boundaries vary significantly. Tenure boundaries in British Columbia may be found as close as 10 meters from the perimeter of netpens or as far away as 300 meters. A requirement that environmental degradation not extend beyond 25 meters from the site boundary establishes an artificial zone of allowable effect that is not biologically significant. My recommendation is that NOSB require that products certified as *organic* be from farms documented to be in full compliance with these various regulations such as is stated in 205.255.(i). These distances are generally measured from the netpen's perimeter and are designed to insure that adverse effects occur only locally and do not become widespread or long-lasting.

6.0. Summary. There are costs associated with every form of food production. Certainly the loss of topsoil at rates that are 16 to 300 times faster than it is being replenished in association with the production of grains needed to bake loaves of bread and other crops is not sustainable. Wild stocks of fish are being depleted in an effort to supply humankind's demand for aquatic protein. Almost none of these more traditional ways of producing food have received the scrutiny that aquaculture has. For instance, what are the long-term costs associated with soil loss around the world? What are the environmental costs (as defined in this paper) associated with derelict (lost) fishing nets and pots? What is the overall environmental cost of taking 3.5 to 4.5 million metric tons of Pollack out of the North Pacific each year? How does this reduction in the standing biomass affect other parts of the food chain including benthic communities? The scrutiny of these issues is so low that no literature was found quantifying lost fishing gear, let alone the environmental cost in terms of fish and shellfish that dies in these traps each year. In this respect, some of the current emphasis on eliminating environmental effects associated with aquaculture is akin to Nero playing his fiddle while Rome was burning. The path to sustainability can only be achieved through a holistic and scientifically rigorous approach to managing earth's resources. A systematic approach to these assessments requires the following:

- An acknowledgement that there are environmental costs associated with all forms of food production;
- Identification of the direct and indirect environmental costs associated with all forms of food production;
- Prioritization of the identified costs of food production;
- Focused research to minimize (not necessarily eliminate) costs associated with the least sustainable production methods.

Regional nature of costs. It should be emphasized that environmental responses depend not only on the hazards associated with food production, but also on specific environments. For instance, adding nutrients to open Northeast Pacific ocean water does result in a significant response.

Adding the same amount of nutrient in another region or in closed estuaries in the Northeast Pacific might result in significant effects.

Identifying real effects versus effects "per se." An assessment that allows quantification of actual effects rather than *effects per se* associated with food production is needed. That requires development of an understanding of the environmental response to the agricultural activity. For instance, the discharge of nutrients in water from shrimp culture ponds is an *effect per se.* The environmental cost of that effect requires an understanding of background nutrient concentrations in the receiving water and other conditions (turbidity, light availability, etc.) affecting primary productivity. Direct measurement of natural productivity is the most direct and sensitive way of measuring environmental costs. Surrogate endpoints, such as free sulfides and redox potential are far less time consuming and expensive than macrofaunal community assessments. However, these are effects *per se* until quantitative cause and negative affect relationships with valuable resources are demonstrated. For instance, the discharge of nutrients can have positive or negative effect on primary production and unless the actual response is understood, it is not possible to assess the cost with confidence.

Uncertainty associated with environmental cost assessments. As shown in this paper, several effects associated with Category II hazards are reasonably well understood and based on empirical evidence. Other Category II hazards, such as the environmental response to exceedances of an ecosystem's *carrying capacity* are less well understood. The environmental response to many, if not most, Category IV hazards are not well understood and cannot be quantified. Development of the understanding required to quantify environmental costs typically requires years of effort and significant investment of resources. In the absence of empirical data, estimates must be made on qualitative determinations, which increase uncertainty in cost assessments (Figure 25). The point is that quantifying the costs associated with various food producing sectors will take decades and future investments in research. In these instances, people and organizations interested in sustainable food production can either throw up their hands in frustration or they can make best use of the information and experience available to complete the assessments. This is conceptually illustrated in Figure 25. The advantage of this approach is that it allows at least a qualitative understanding of the costs of food production and it allows us to focus our energy on mitigating the most pressing costs.

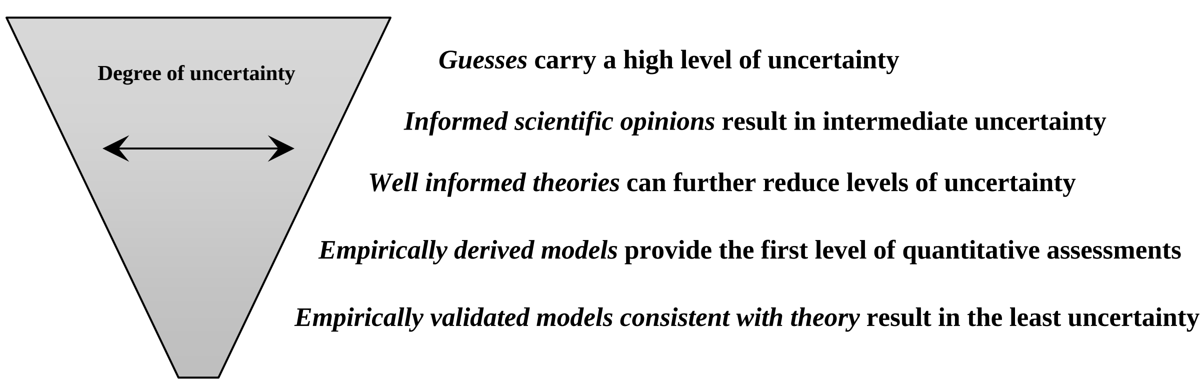

Figure 25. Relationship between the uncertainties associated with cost comparisons between food producing sectors and the uncertainty of the assessments.

Transparency. It is important that the identification and prioritization of environmental costs associated with food production be conducted in a transparent manner. That implies acknowledging, and where possible quantifying, the uncertainty described in Figure 25. Every report assessing the environmental costs associated with food production should include an acknowledgement of the costs included in the assessment and those that are excluded.

The body of this report has focused on the effects associated with organic enrichment from salmon aquaculture to illustrate the level of detail and years of work necessary to understand just one facet of the costs associated with a single hazard. Understanding these costs is not a trivial pursuit. However, achieving sustainable use of earth's resources is an important goal that must be undertaken in a systematic way if future generations will not look back at the 21st century and condemn us for unwise use and management of these resources. A beginning can be made by bringing together multidisciplinary teams of scientists to define the scope and context of the problem. Such an effort will help guide existing and future work to focus on the most pressing problems. This need and approach is frequently cited and often discussed. Unfortunately programs to accomplish it are infrequently, if ever, implemented.

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